Viability Criteria for Application to Interior Columbia Basin Salmonid ESUs

Review Draft
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Interior Columbia Basin Technical Recovery Team

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Table Of Contents

Background

One of the main tasks assigned to Technical Recovery Teams (TRTs) is the establishment of biological viability criteria for application to Evolutionarily Significant Units (ESUs) of salmon and steelhead listed under the Endangered Species Act[1]. A viable ESU is self-sustaining in nature, not only numerically persistent over time, but also is functional in both the ecological and evolutionary states (McElhany et al., 2000, ISAB 2005). Biological viability criteria are quantitative metrics that describe ESU characteristics associated with a low risk of extinction for the foreseeable future. These biological viability criteria are intended to inform long-term regional recovery planning efforts, including the establishment of delisting criteria. The Interior Columbia Basin Technical Recovery Team (ICTRT) developed a set of viability criteria and guidelines specific for Interior Columbia Basin listed ESUs; those viability criteria are described in this paper.

Our ESU level viability criteria consider the appropriate distribution and characteristics of component populations in order to maintain the ESU in the face of long-term ecological and evolutionary processes. The viability criteria were based on guidelines in the NOAA Technical Memorandum *Viable Salmonid Populations and the Recovery of Evolutionarily Significant Units* (McElhany et al. 2000), the results of previous applications (Puget Sound TRT, 2004 and Lower Columbia/Willamette TRT, 2003 & 2006) and a review of specific information available relative to listed Interior Columbia ESU populations. The population level viability guidelines provided in McElhany et al. (2000) are organized around four major parameters: abundance, productivity, spatial structure and diversity. Our population level viability criteria are designed to address, in combination, all four of these key parameters. Since we defined our ESU level viability criteria in terms of the viability of component populations, we were able to relate ESU viability directly to the primary drivers of evolutionary and ecological functionality.

The Interior Columbia Technical Recovery Team

The ICTRT is one of a series of Technical Recovery Teams established by the National Marine Fisheries Service (NOAA Fisheries) to provide scientific input into regional recovery planning efforts for listed salmon and steelhead. The TRTs are chaired by scientists from Northwest Fisheries Science Center or the Southwest Fisheries Science Center and include experts in population dynamics, conservation biology, ecology and other disciplines relevant to recovery planning. TRT members include scientists from federal and state agencies, tribal resource divisions, academia, and private consultants.

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[1] NMFS has recently delineated steelhead only distinct population segments (DPS) for West Coast steelhead (71 FR 834). In this report we use the generic term ESU to refer a steelhead DPS.

Applications of Viability Criteria

The biological viability criteria described in this report were explicitly developed to inform long-term regional recovery planning efforts and delisting criteria. Given that intent, we worked to express the criteria in objective, relatively specific, and measurable metrics. The quantitative specificity of the criteria gives conservation planners a clear picture of the attributes of viable populations, MPGs and ESUs, while providing a level of transparency that facilitates critical review and future refinements. However, we recognize that there are local circumstances that may make the criteria less applicable to particular populations and that there is uncertainty in both the data and the criteria themselves. For this reason, we have left some room for interpretation or modification of the criteria when well-documented and justified circumstances exist.

The Endangered Species Act (ESA) requires that recovery plans for listed species contain “measurable and objective criteria” that when met would result in the removal of the species from the endangered species list. To be removed from the list, a species must no longer be in danger of or threatened with extinction. Court rulings and NMFS policy indicate that delisting criteria must include both biological criteria and listing factor criteria that address the threats to a species (i.e., the listing factors in ESA section 4[a][1]). The viability criteria relate most directly to the biological delisting criteria; however, they are not synonymous. NMFS establishes delisting criteria based on both science and policy considerations. For instance, science can identify the best metrics for assessing extinction risk and thresholds of those metrics associated with a given level of risk, but setting the acceptable level of risk for purposes of the ESA is a policy decision.

The ICTRT criteria were developed with explicit recognition that the ultimate choice of an acceptable risk level in recovery planning is a policy choice. The ICTRT population level viability criteria are expressed relative to an acceptable risk level of a 5% probability of extinction in a 100-year period. This level of risk is consistent with VSP guidelines (McElhany et al., 2000), the conservation literature (e.g., NRC, 1995), and previous policy guidance that biological objectives based on a 5% (or less) risk of extinction over a 100 year period provide adequate benchmarks for use in assessing recovery.(NMFS, 2005). In addition, we recognize that recovery plans may use these basic biological criteria as a path for setting broad-sense recovery goals for an ESU that reflect policy needs to address additional societal values such as providing for fully functioning ecosystems, fishing opportunities and opportunities for the public to appreciate salmon in the wild. Additional policy guidance on relative to recovery planning applications of TRT products can be found on the following website: http://www.nwr.noaa.gov/Salmon-Recovery-Planning/ESA-Recovery-Plans/Other-Documents.cfm

In addition, the criteria we used to express viability facilitate the development of effective recovery strategies by focusing attention on specific, often spatially explicit, biological conditions or processes. For example, our criteria include quantitative metrics expressed in terms of the current distribution of spawners relative to spatially explicit maps of historical production potential within a population. We provide examples of the relative risk associated with a range of general spawning area configurations. The descriptions of risk associated with alternative configurations provide recovery planners with an objective basis for targeting actions to address that component of viability. Our abundance and productivity criteria were designed to be used, in combination with current assessments, to inform recovery planning efforts as to the relative magnitude of changes in survival and habitat capacity needed to achieve viable status. They can also provide insight into whether productivity alone, or both productivity and capacity might need to be improved. Current status reviews developed by the ICTRT with input from regional technical teams will be compiled in a separate ICTRT document. We have included two current population status assessments with this report to illustrate application of the ICTRT viability criteria. Additional draft assessments are available at our website: http://www.nwfsc.noaa.gov/trt/trt_current_status_assessments.cfm.

Definitions

To understand the scope and focus of this report, it is useful to start with some definitions. The ESU and Population viability sections also include definitions of key terms and concepts. These definitions are intended to be consistent with current NMFS definitions and policy.

Biological viability criteria – Viability criteria are the primary focus of Part 1 of this report. Viability criteria describe biological or physical performance conditions that when met indicate a population or ESU is not likely to go extinct. Viability criteria have two components: a ***metric***, which is the parameter measured, and the ***criteria***, which are the values of the metric at which risk levels for a population or ESU are assigned. Viability criteria focus on the biological performance of the fish as the primary indicator of extinction risk. Viability criteria are intended to inform delisting criteria and therefore focus on metrics that can be used in current and future status evaluations. In 2005, NOAA published a policy in the Federal Register clarifying the role of hatchery production in risk assessments (70 FR 123: 37204). As currently being applied, the policy states that a non-listed ESU must be naturally self-sustaining and must be able to persist without input of hatchery-produced fish. The viability criteria described in this report are consistent with that standard.

Current status evaluation – A current ESU or population status evaluation is an assessment of the current extinction risk for populations and ESUs. Like viability criteria, current status evaluation relies on ***metrics and thresholds.*** However, viability criteria (as defined above) differ in an important way from current status evaluations. Current status evaluations are based on the information that is currently available on the ESU or population in question, whereas viability criteria describe those conditions under which populations might be considered to have a particular level of risk.

ESU scenario – The viability criteria described in this report allow for some flexibility in which populations will be targeted for a particular recovery level to achieve a viable ESU. An ESU scenario is an explicit description of which populations in an ESU are targeted for a given recovery level. Developing an ESU scenario requires both biological and policy considerations.

Relationship to Previous ICTRT Reports

Previous drafts of the ICTRT viability criteria were made available to provide guidance to regional recovery planning efforts that were ongoing concurrently with the development of these viability criteria. Early versions of the criteria were tested on some populations and refined based on lessons learned from the tests and input from regional recovery planners. We also have addressed technical peer review comments generated as a result of these early applications. The specific set of objectives and the particular measures associated with each component of our criteria have not changed. In some cases, the definition of certain risk levels in terms of a particular metric have been modified to facilitate more objective and consistent application of the criteria as well as to reflect new or better information as it became available. In addition, updates to the analyses used to estimate historical production capacity have resulted in changes in the assignment of some populations to a historical size category.

Considering Uncertainties

We recognize that uncertainty is an important consideration in setting risk criteria for natural populations. We considered categories of uncertainties in developing viability criteria for Interior Basin ESUs. First, some of our knowledge of the biological structure and functioning of specific ESUs is based on statistical sampling. Estimates of particular parameters are therefore subject to sampling variability. We provide results from sensitivity analyses to illustrate the potential effect of key uncertainties associated with several of our quantitative criteria. We encourage the use of multiple models or lines of evidence in assessing risk. Second, we provide options for directly incorporating a measure of uncertainty in evaluating current status. In addition, we identify topics for further scientific evaluation that could decrease uncertainties or lead to future improvements in particular criteria. Lastly, our criteria incorporate current understanding of environmental processes and their links to population dynamics. We encourage consideration of alternative future scenarios in developing strategies to achieve viability.

Organization

This report is organized into four sections. The initial section includes a general description of ESU hierarchical structure. The second section describes our ESU and Major Population Group (MPG) level criteria. The third section describes our population level criteria, including general examples and guidelines for using the criteria to determine the relative viability of a population. It also presents a method for generating an aggregate population risk rating and a discussion of approaches for addressing uncertainty in population viability metrics. The fourth section includes a summary of opportunities to improve or validate key assumptions through further monitoring and evaluation as well as a summary regarding application of the criteria described in this report. Appendices and attachments are included that provide more detailed technical analyses used in developing some of the population viability criteria, describe potential combinations of populations to achieve ESU viability and the role of repopulating extirpated areas in ESU viability, and provide some examples of application of population viability criteria.

Hierarchical Levels for Estimating ESU Viability

The ICTRT viability criteria reflect the hierarchical structure of Interior Columbia Basin ESUs (McElhany et al. 2000). In a previous ICTRT report, we described the structure of each Interior Columbia listed ESU in terms of discrete populations organized into Major Population Groups (Figure 1). Populations have been formally defined as a group of individuals that are demographically independent from other such groups over a 100-year time period (McElhany et al. 2000). We define Major Population Groups (MPGs) as sets of populations that share genetic, geographic (hydrographic), and habitat characteristics within the ESU (ICTRT 2003, 2005). They are analogous to "strata" as defined by the Lower Columbia-Upper Willamette TRT and "geographic regions" described by the Puget Sound TRT.

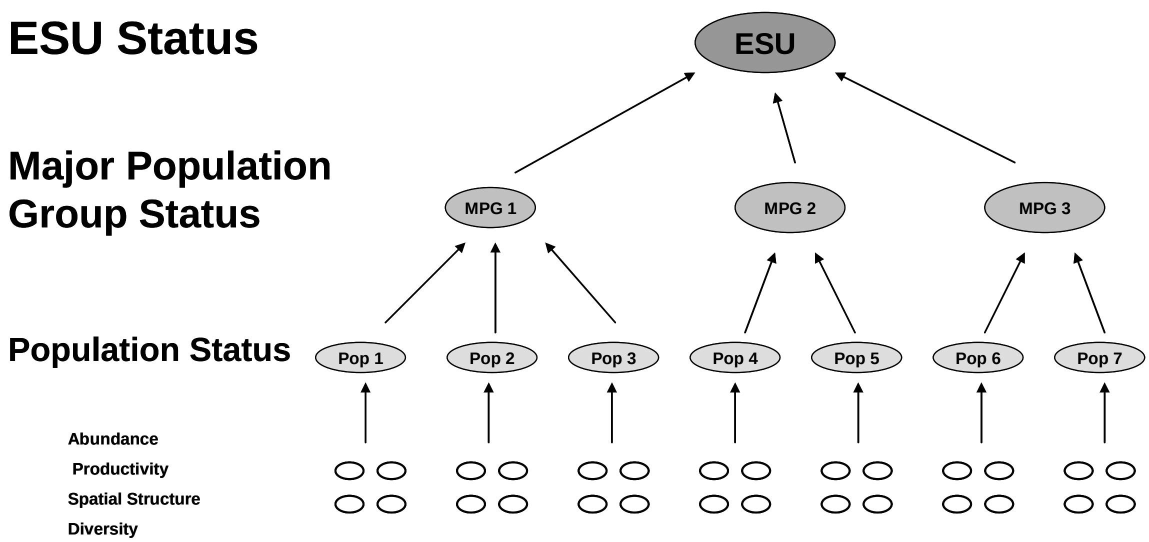


Figure 1. Diagram illustrating the hierarchy of ESU, MPG, and population level viability criteria.

At the population level, our viability criteria are expressed in terms of four attributes; abundance, productivity, spatial structure and diversity. The ICTRT designated major spawning areas (MaSAs) and minor spawning areas (MiSAs) as a framework for expressing within population spatial structure and diversity criteria (Appendix C).

Populations identified by the ICTRT range widely in terms of total tributary drainage area and complexity. Examples of populations occupying smaller drainages include Asotin Creek and Sulphur Creek (Snake River Steelhead and Spring/summer Chinook ESUs); Rock Creek and Fifteen Mile Creek (Middle Columbia ESU); and the Entiat River (Upper Columbia Steelhead and Spring Chinook ESUs). Populations using relatively large, complex tributaries include Upper John Day steelhead, Wenatchee and

Methow River Spring Chinook; and Lemhi River steelhead and spring/summer chinook. This natural variation in size and complexity suggests that even historically, populations likely varied in their relative robustness, or resilience to perturbations. Because of this variation, the TRT did not adopt a "one-size-fits-all" approach to population-level criteria. Considerations for relative population size and complexity characteristics are reflected in the population and Major Population Grouping viability criteria developed by the ICTRT. We provide population specific estimates of the amount and complexity of tributary spawning habitats in the Population Viability Criteria section of this report.

ESU/MPG Viability Criteria

McElhany et al. 2000 identifies three factors to consider in assessing the viability of an ESU in terms of its component populations: 1) catastrophic events, 2) long-term demographic processes and 3) long-term evolutionary potential. Catastrophic events are localized, relatively sudden impacts that can severely reduce or eliminate a population. The potential for these types of events impacting a particular population are not usually captured in short-term (e.g., 10 to 100 year) assessments of annual environmental variations. Long-term demographic processes relate to the potential for recolonization of locally extirpated populations within an ESU from other populations. Evolutionary potential of an ESU relates to the role diversity plays in ESU viability. Both of these processes operate on time scales extending out to hundreds of years.

ICTRT ESU Criteria

The major objectives of our ESU/MPG level viability criteria are to ensure preservation of basic historical metapopulation processes including: 1) genetic exchange across populations within an ESU over a long time frame; 2) the opportunity for neighboring populations to serve as source areas in the event of local population extirpations; 3) populations distributed within an ESU so that they are not all susceptible to a specific localized catastrophic event. To meet these objectives a viable ESU will likely have some populations meeting viability standards close to each other AND some populations meeting viability standards relatively distant from each other (McElhany et al. 2000, Isaak et al. 2003).

A variety of recovery scenarios may lead to a viable ESU. Different scenarios of ESU recovery may reflect alternative combinations of viable populations and specific policy choices regarding acceptable levels of risk. The particular recovery objectives for Interior Columbia ESUs will be generated by policy and technical interactions in conjunction with regional planning efforts. We provide the following criteria to describe the biological characteristics of a viable ESU to inform the development of specific recovery objectives for Interior Columbia ESUs.

Our ESU-level viability criterion is:

All extant MPGs and any extirpated MPGs critical for proper functioning of the ESU should be at low risk.

We express our ESU viability criterion in the context of Major Population Groups (MPGs)—geographically and genetically cohesive groups of populations within an ESU that are thus critical components of ESU-level spatial structure and diversity. Historically, these groupings of populations within an ESU likely functioned as metapopulations—formally defined as sets of discrete, largely independent populations whose dynamics are driven by local extinction and with limited interbreeding and recolonization among populations (after Levins,1969). We do not have sufficient information on movement or exchange rates among Interior Columbia Basin populations to directly model MPGs or ESUs as metapopulations. We have defined MPG-level viability criteria to ensure robust functioning at the metapopulation level and mitigate the risk of catastrophic loss of one or more populations. MPG viability depends on the number, spatial arrangement, and diversity associated with its component populations. Criteria for evaluating the relative viability of a population are provided in the following section of this report.

We have developed the following MPG-level criteria considering relatively simple and generalized assumptions about movement or exchange rates among individual populations (details for population viability are provided in the next section).

An MPG meeting the following five criteria would be at low risk:

1. ***At least one-half of the populations historically within the MPG (with a minimum of two populations) should meet viability standards.***

2. ***At least one population should be classified as "Highly Viable."***

3. ***Viable populations within an MPG should include some populations classified (based on historical intrinsic potential) as "Very Large", "Large" or "Intermediate" generally reflecting the proportions historically present within the MPG. In particular, Very Large and Large populations should be at or above their composite historical fraction within each MPG.***

4. ***All major life history strategies (e.g. spring and summer run-timing) that were present historically within the MPG should be represented in populations meeting viability requirements.***

5. ***Populations not meeting viability standards should be maintained with a) sufficient productivity so the overall MPG productivity does not fall below replacement (i.e. these areas should not serve as significant population sinks) and b) sufficient spatial structure and diversity demonstrated by achieving Maintained standards.***

The ICTRT ESU/MPG criteria follow the basic guidelines provided in McElhany et al. 2000. The specific rationale for the individual components of our MPG/ESU level criteria are described below.

Minimum Number of Viable Populations

Modeling efforts incorporating spatial structure, local and correlated catastrophes and dispersal suggest that extinction risk of a metapopulation as a whole decreases rapidly as additional viable populations are added to the group (Ruckelshaus et al. 2003, 2004, Tear et al. 2003). Kendall et al. (2001), in conducting a PVA of Gila Trout, found that extinction risk was highly sensitive to the number of populations included in the model. Rieman and Dunham (2000) and Fagan (2002) discuss the importance of metapopulation structure to overall risk for fish populations occupying dendritic habitats as well as the associated difficulties in accurately modeling particular situations. Based on these analyses, we generally conclude that an MPG containing only one viable population would be at substantially greater risk of extirpation than one with two or more populations, and that additional populations present within an MPG would further decrease the risks to the functioning of the MPG.

We recommend that a minimum of one-half of the populations historically present (but no less than 2) within an MPG be viable based on two major consideration. First, having multiple viable populations can provide a spatial distribution that provides for normative dispersal and gene flow among populations while still supporting within-MPG diversity. Second, because populations that are close to each other are more likely to have some demographic linkage (Bentzen et al. 2001), having multiple viable populations reduces extinction risk due to local catastrophic events. Reducing extinction risk related to catastrophic events typically requires a reasonable proportion of the populations within the MPG. Connectivity among populations in the MPG is expected to increase as the number of viable populations increases and distances between proximate populations decreases. Kendall et al. (2001) linked increased connectivity to increased recolonization of populations subject to catastrophic losses and improved viability of Gila trout. We expect this same principle applies to the metapopulation-like structure of an MPG and increased viability of the MPG is achieved by having multiple viable populations. An objective for the combinations of Viable and Maintained populations required to meet our MPG criteria is achieving a composite MPG productivity at or above replacement, thus ensuring long-term persistence of the ESU (Holmes and Semmens, 2004, Gunderson et al. 2001).

Achieving viability goals for the minimum number of populations will likely require attempting to meet those targets in more than just those populations because the efficacy of recovery efforts is uncertain. For example, if there is an 80% chance that recovery will be successful in each of a set of three populations identified, there is an overall 51% probability of recovering three populations if recovery efforts are limited to those three populations (McElhany et al. 2003). To have more than a 95% probability of recovering three populations in this case would require attempting recovery of six populations. Consequently, more populations than the minimum should be targeted for viability. This strategy would also address the uncertainty inherent in the assumption that 2 or half of the populations in an MPG are adequate for viability.

Include Highly Viable populations

The ICTRT recommends that at least one population within each MPG should be Highly Viable, following the recommendation in McElhany et al. (2000) The presence of highly viable populations distributed across the ESU provides source populations that can recolonize populations that have experienced catastrophic losses (McElhany et al. 2000; Gunderson et al. 2001). Also, achieving a higher level of viability for a subset of populations scattered across the ESU provides some protection against future environmental conditions substantially deviating from historical patterns.

Population Sizes Represented

We include recommendations for the size distribution of populations within an MPG for a similar reason—large populations are more likely to have served historically as "source" areas for the group of populations (McElhany et al. 2000) In addition, larger populations almost always consist of 2 or more relatively discrete production areas, each of which was capable of sustaining 500 or more spawners. From the perspective of localized catastrophic risks, these populations are at lower risk of total loss for a brood cycle or longer than populations confined to a single sub watershed or mainstem reach. An MPG consisting of small populations at low risk and large populations at relatively higher risk is likely to be at higher risk overall than one that includes large populations in a low-risk condition.

Major Life History Patterns Represented

Major life history variations (e.g., spring vs. summer adult run timing and the associated differences in spawning timing/areas) represent an important component of the diversity within an ESU. These major life history patterns represent adaptations to the range of environmental conditions experienced by populations across the historical range of an ESU. Requiring the security of low-risk levels for at least one population representing each historical life history variation within an MPG provides a basis for the ESU to adapt to future conditions.

Maintained Populations

Our criteria focus efforts at recovering a minimum number of populations within each MPG to viable levels. In many cases there will be one or more additional extant populations within an MPG. The ICTRT established the maintained criterion for application to these populations. The primary intent is to avoid situations where one or more of these populations serve as an overall 'sink' for production across an MPG. In addition, meeting the maintained criterion for these populations contributes to connectivity within and among MPGs and promotes the preservation of genetic and life history diversity. The Population Viability Criteria section below includes a discussion of objectives and criteria for maintained populations. This recommendation is analogous to the element of the Lower Columbia/Willamette TRT viability criteria that stipulates that populations not meeting viability criteria be maintained at a levels providing ecological and evolutionary function to the ESU as a whole (McElhany et al. 2003).

Combined Effects of Meeting MPG Criteria

Having all MPGs within an ESU at low risk addresses the three ESU level considerations identified by McElhany et al. 2000. Protection against long-term impacts of localized catastrophic loss is gained by the presence of multiple, relatively nearby viable and maintained populations to serve as a source of re-colonization. MPGs were defined, in a large part, based on genetic and ecological differentiation. For example, Figure 2 illustrates the range in elevation associated with historical spawning reaches for Snake River Spring-Summer Chinook ESU populations. Annual temperature and precipitation patterns are substantially influenced by elevation. The ICTRT criteria requiring viable populations in each of the five extant MPGs of this ESU would result in sustainable production across a substantial range in environmental conditions. The presence of viable populations across MPGs would preserve a high level of ESU diversity, thereby promoting long-term evolutionary potential for adaptation to changing conditions. This criterion is also consistent with recommendations for other ESUs in the Pacific Northwest (e.g., McElhany et al. 2006, PSTRT, 2002).

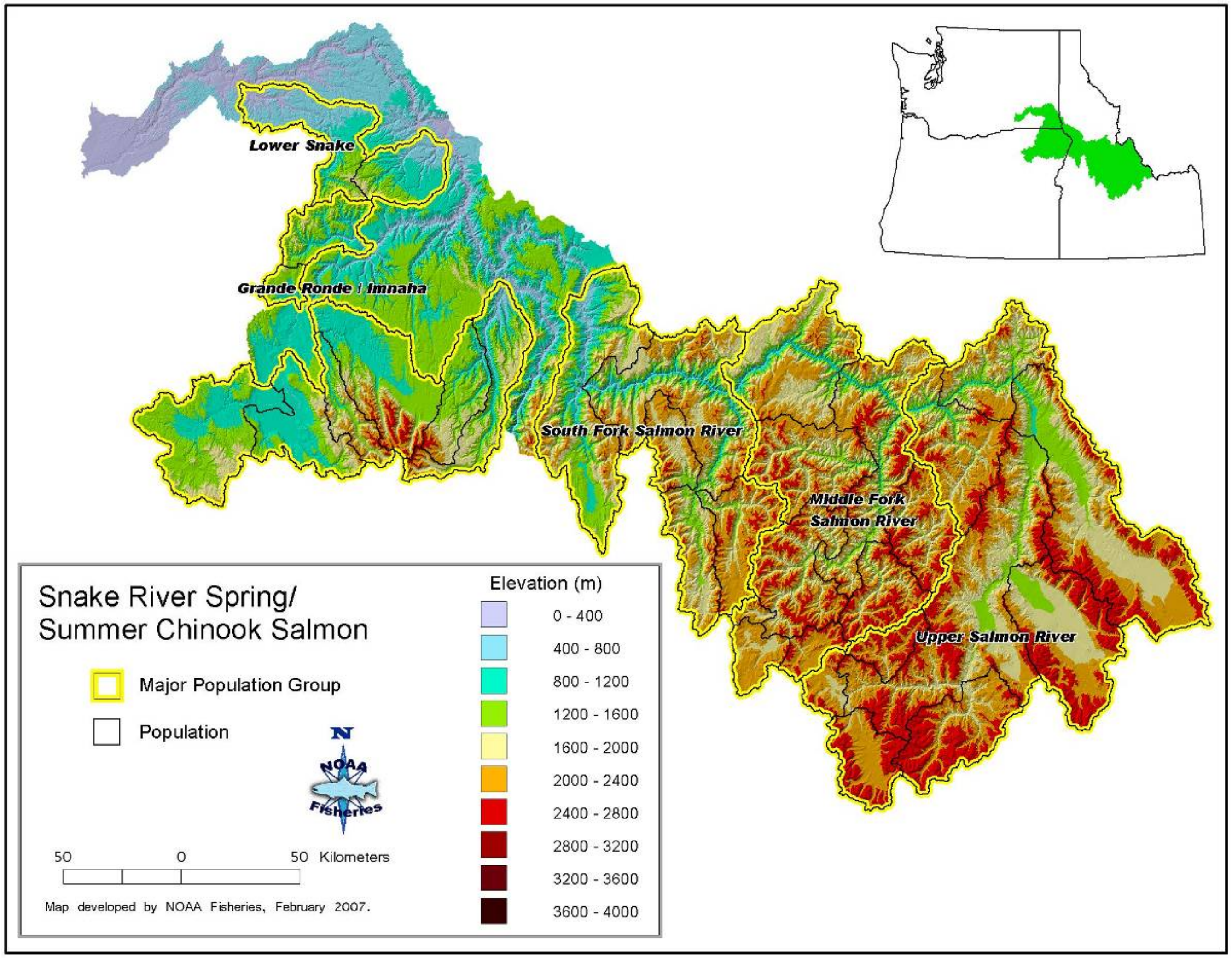


Figure 2.a. Snake River Spring/Summer Chinook ESU distribution of populations and Major Population Groups (MPGs) relative to elevation.

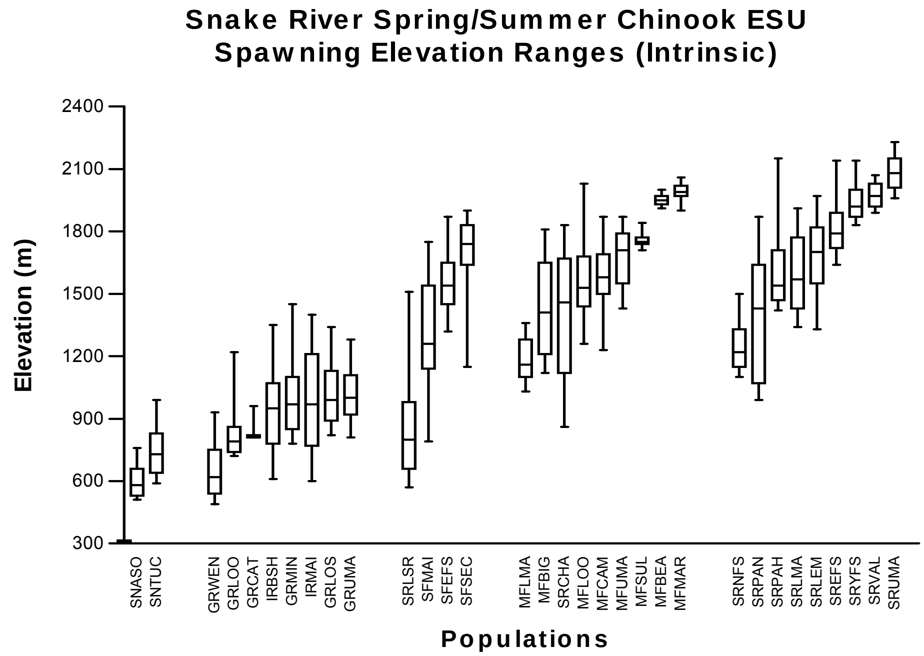


Figure 2.b. ESU Snake River Spring/Summer Chinook population median elevations. Boxes represent the range of elevation in the middle 50% of the population, while bars represent the middle 90%.

ESUs with a single MPG

ESUs that contain only one MPG are inherently at greater extinction risk due more limited spatial structure and diversity and potentially abundance and productivity. In addition, they typically have fewer component populations, which also increases risk level (Boyce 1992, Tear et al. 2003). We provide more stringent criteria for ESUs with a single MPG than ESUs with multiple MPGs to mitigate this inherently higher risk.

ESUs that contained only one MPG historically or that include only one MPG critical for proper function should meet the following criteria:

- A single MPG should meet all the requirements to be at low risk (see above). In addition:

  1. Two-thirds or more of the historical populations within the MPG should meet viability standards; AND

  2. At least two populations should meet the criteria to be "Highly Viable."

Extirpated MPGs

The ICTRT has conducted an evaluation to determine whether extirpated MPGs are critical for proper functioning of the ESU (see Attachment 1). The evaluation was based on the following general considerations:

- Likely demographic (abundance and productivity) contribution of the MPG and its component populations to the ESU.

- Spatial role of the MPG in the ESU (e.g. does the extirpated MPG create a gap in the distribution of the ESU?)

- Likely contribution to overall ESU diversity (e.g. does the extirpated MPG occupy habitats that are substantially different from other habitats currently occupied in the ESU?)

Alternative Recovery Scenarios

Three of the listed Interior Columbia ESUs include four or more MPGs (major population groups) each of which contains multiple extant populations (Tables 2a-c). In those circumstances, there can be several different viable population scenarios at the MPG level that would meet the ICTRT viability criteria. We have summarized potential viability scenarios for each ESU in a ICTRT memo (Attachment 1). In addition, the role of large extirpated areas on the overall risk for an ESU varies with the characteristics of the currently accessible areas. We treat the likely changes in risk that would result from the establishment of self-sustaining populations in these extirpated areas in a second memo (Attachment 2).

Population Level Viability Criteria

Here, we describe the criteria for use in assessing viability at the individual population level. We have grouped specific population level criteria into two basic subsets; measures addressing abundance and productivity and a set reflecting spatial structure/diversity elements. We also present a framework for compiling an aggregate risk score for a population based on the results of applying the individual criteria.

Historical Populations: Size and Complexity

Populations of listed stream type chinook salmon and steelhead within the Interior Columbia River vary considerably in terms of the total area available to support spawning and rearing. The ICTRT developed a method for adapting viability curves to reflect estimates of the historical amount of potentially accessible spawning and rearing habitat available to a specific population. A more detailed description of the approach is provided in Appendix B. The measure of historical habitat we used is primarily driven by spawning habitat considerations. We emphasize spawning population size in these viability assessments because of the direct link to population genetic characteristics, demographics, etc. The same habitat characteristics we used in the assessment generally reflect relative juvenile production potential, but we recognize that an analysis focused on estimating the relative amount of juvenile habitat would recognize additional combinations of habitat characteristics. Analyses aimed at evaluating limiting factors or the potential effects of proposed habitat actions should consider juvenile rearing habitat. We initially focused on an application for stream type chinook and steelhead populations because of the availability of representative data sets and the relative number of listed ESU populations. We adapted the approach to accommodate the biological characteristics and available data for Snake River Fall Chinook and Snake River Sockeye populations, respectively.

Estimating Historical Capacity

In summary, a measure of the historic spawning/rearing area for each population was generated using a simple model of historical intrinsic potential. That model is driven by estimates of stream width, gradient, valley width, and confinement derived from a GIS-based analysis of the tributary habitat associated with each population. Additional screens were added for steelhead intrinsic potential that included sediment, soil erodibility and flow velocity. Each accessible 200 meter reach within the tributary habitat associated with a specific population was assigned an intrinsic productivity rating based on the particular combination of physical habitat parameters listed above. A weighted estimate of the total amount of rated habitat historically available to each population was generated. The habitat ratings for each potential spawning reach were assigned a relative weighting and summed by population

We established a set of four population size categories (Basic, Intermediate, Large and Very Large) for Interior basin stream type chinook and steelhead populations. For each species, populations were ordered and grouped according to the estimated amount of historical spawning/rearing habitat (Appendix B). Two considerations were used to determine breakpoints between category assignments: median size of populations within a putative grouping and the occurrence of relatively large incremental differences between adjacent populations in the species sequences. The smallest populations were grouped into the Basic size category. Populations assigned to the Basic size categories tended to be simple in complexity, often with a relatively linear arrangement of spawning/rearing reaches. Median population size roughly doubled between size categories. Populations with significantly higher amounts of potential spawning habitat usually exhibited a higher degree of spatial diversity—e.g., multiple tributary branches. Contemporary redd survey results indicated that the distribution of spawners across sub areas within a population was likely to be patchy. Relatively high spawning concentrations in particular subareas could be achieved in the larger, more complex population at lower overall spawning densities. Size category assignments for the specific populations within each of the listed Interior Columbia ESUs are provided in Tables 2a-e. Relative population size estimates for Snake River Fall chinook and sockeye populations are described in Appendix B.

Population Spatial Complexity

We used two methods to characterize the relative within-population complexity of tributary spawning habitats—assigning each population to one of four general structural complexity categories (Table 1), and estimating the number of relatively large, contiguous production areas within each population (Appendix C). We hypothesize that the increased protection against catastrophic loss provided by multiple large spawning areas within a single population would be analogous to the risk reduction associated with having multiple independent populations within an ESU. We defined an empirical, data-based measure of potential spawning habitat as a baseline for our criteria. We defined a branch as a river reach containing sufficient habitat to support 50 spawners. Major spawning areas (MaSAs) were defined as a system of one or more branches that contain sufficient habitat to support 500 spawners. For spring/summer chinook, this value was 100,000m^2, and for steelhead it equaled 250,000m^2. We generated aggregation values by using hydrology tools within GIS (see Appendix C). We defined contiguous production areas capable of supporting between 50 and 500 spawners as minor spawning areas (MiSAs).

Table 1**.** Population spatial complexity designations

Category	Description
A.	Linear structure, with no more than 2 branches in one major spawning area. Typically small (basic) drainages.
B.	Dendritic tributary structure including 2 or more major spawning areas. Typically intermediate or large drainages.
C.	Trellis-structured drainage including mainstem spawning and multiple branches.
D.	Populations with one or more major spawning areas with well-separated minor spawning areas downstream.

Stream Type Chinook and Steelhead Populations

Each population was assigned to a size category based on the total amount of weighted spawning habitat and given a complexity rating based on the estimated relative distribution of historical spawning habitat (Tables 2a-e).

Table 2.a. Intrinsic size and complexity ratings for **extant Snake River Spring Chinook ESU** populations organized by Major Population Groupings. Complexity categories: A = linear; B=dendritic; C= trellis pattern; D= core drainage plus adjacent but separate small tributaries. Underlined entries represent a change from the previous designation. Size categories in parentheses represent core tributary production areas.

Major Population Group	Population	Weighted Area Category	Complexity Category	Complexity #MaSA (#MiSA)
Lower Snake	Tucannon R	Intermediate	A	1 (0)
	Asotin R. (ext)	Basic	A	0 (1)
Grande Ronde/Imnaha R	Lostine/Wallowa R.	Large	B	3 (1)
	Upper Grande Ronde R.	Large	B	3 (2)
	Catherine Creek	Large	B	2 (2)
	Imnaha R. Mainstem	Intermediate	A	1 (1)
	Minam R.	Intermediate	A	2 (0)
	Wenaha R.	Intermediate	A	1 (0)
	Big Sheep Cr. (ext)	Basic	A	0 (1)
	Lookingglass Cr. (ext)	Basic	A	0 (1)
South Fork Salmon	South Fk Mainstem	Large	C	2 (2)
	Secesh R.	Intermediate	A	1 (1)
	East Fk/Johnson Cr.	Large	B	2 (0)
	Little Salmon R.	Inter. (Basic)	D	0 (3)
Middle Fork Salmon	Big Creek	Large	B	3 (0)
	Bear Valley	Intermediate	C	3 (0)
	Upper Mainstem MF	Intermediate	C	1 (2)
	Chamberlain Cr.	Inter. (Basic)	D	1 (3)
	Camas Creek	Basic	B	1 (1)
	Loon Creek	Basic	C	1 (0)
	Marsh Creek	Basic	C	1 (0)
	Lower Mainstem MF	Basic	A	0 (1)
	Sulphur Creek	Basic	A	1 (0)
Upper Salmon	Lemhi	Very Large	B	3 (2)
	Lower Mainstem	Very Large	C	3 (5)
	Pahsimeroi	Large	B	5 (0)
	Upper Salmon East Fk	Large	C	1 (0)
	Upper Salmon Mainstem	Large	C	3 (0)
	Valley Cr.	Basic	A	1 (0)
	Yankee Fork	Basic	C	1 (0)
	North Fork Salmon R.	Basic	D	1 (0)
	Panther Cr. (ext)	Intermediate	C	1 (2)

Table 2.b. Intrinsic size and complexity ratings for historical **Snake River Steelhead ESU** populations organized by Major Population Groupings. Complexity categories: A = linear; B=dendritic; C= trellis pattern; D= core drainage plus adjacent but separate small tributaries. Size categories in parentheses represent core tributary production areas.

Major Population Group	Population	Weighted Area Category	Complexity	
			Category	#MaSA (#MiSA)
Lower Snake	Tucannon R	Intermediate	A	1 (2)
	Asotin R.	Basic	D	2 (5)
Grande Ronde	Upper Grande Ronde R.	Large	B	6 (7)
	Wallowa River	Intermediate	B	4 (2)
	Lower Grande Ronde R.	Intermediate	B	2 (5)
	Joseph Creek	Basic	B	3 (3)
Imnaha R.	Imnaha River	Intermediate	B	4 (3)
Clearwater R.	Lower Mainstem	Large	B	6 (5)
	Selway River	Intermediate	B	7 (6)
	South Fork	Intermediate	B	3 (4)
	Lochsa River	Intermediate	B	3 (5)
	Lolo Creek	Basic	C	1 (0)
	North Fork (ext)	Large	B	---
Salmon River	Lemhi	Intermediate	B	3 (2)
	Upper Salmon East Fork	Inter. (Basic)	B	2 (1)
	Upper Salmon Mainstem	Intermediate	B	5 (2)
	Upper Middle Fork	Intermediate	B	6 (3)
	Lower Middle Fork	Intermediate	B	5 (2)
	Chamberlain Cr.	Basic	D	1 (5)
	Pahsimeroi River	Intermediate	C	3 (2)
	Panther Cr	Basic	D	1 (3)
	Little Salmon River	Inter. (Basic)	D	1 (4)
	South Fork	Intermediate	B	3 (4)
	Secesh R.	Basic	C	1 (1)
	North Fork	Basic	D	1 (1)
Hells Canyon Tributaries	Wild Horse/Powder R.	Note: Core spawning areas for this population are blocked to anadromous migration.		

Table 2.c. Intrinsic size and complexity ratings for historical populations within the **MIDCOLUMBIA RIVER STEELHEAD ESU**. Organized by Major Population Groupings. Complexity categories: A = linear; B=dendritic; C= trellis pattern; D= core drainage plus adjacent but separate small tributaries. Size categories in parentheses represent core tributary production areas.

Major Population Group	Population	Weighted Area Category	Complexity	
			Category	# MaSA (# MiSA)
Eastern Cascades	Deschutes (westside)	Large (Inter.)	B	5 (9)
	Deschutes (eastside)	Intermediate	B	6 (4)
	Klickitat River	Intermediate	B	6 (4)
	Fifteenmile Creek	Basic	C	3 (5)
	Rock Creek	Basic	A	1 (0)
	Crooked River (ext.)	Very Large	B	---
	White Salmon (ext)	Basic	C	---
Yakima River	Upper Yakima River	Large	B	14 (2)
	Naches River	Large	B	8 (2)
	Toppenish River	Basic	B	2 (1)
	Satus Creek	Intermediate	B	3 (4)
John Day River	John Day Lower Mainstem	Very Large	B	13 (22)
	John Day North Fork	Large	B	10 (5)
	John Day Upper Mainstem	Intermediate	B	3 (4)
	John Day Middle Fork	Intermediate	B	4 (2)
	John Day South Fork	Basic	B	3 (0)
Umatilla/Walla Walla	Umatilla River	Large	B	13 (3)
	Walla-Walla Mainstem	Intermediate	B	5 (6)
	Touchet River	Intermediate	A	1 (0)
	Willow Cr. (ext)	Intermediate	B	---

Table 2.d. Intrinsic size and complexity ratings for historical populations within the **UPPER COLUMBIA RIVER SPRING CHINOOK ESU**. Organized by Major Population Groupings. Complexity categories: A = linear; B= dendritic; C= trellis pattern; D= core drainage plus adjacent but separate small tributaries.

Major Population Group	Population	Weighted Area Category	Complexity	
			Category	# MaSA (# MiSA)
Eastern Cascades	Wenatchee	Very Large	B	5 (4)
	Methow	Very Large	B	4 (1)
	Entiat	Basic	A	1 (0)
	Okanogan River (ext) (US portion only) [1]	Intermediate	D	1 (3)

[1] Spring Chinook historically occupied tributary habitat in both the U.S. and Canada. Current ICTRT analyses are focused on the US portion, although additional MSAs or populations may exist in the Canadian portion.

Table 2.e: Intrinsic size and complexity ratings for historical populations within the **UPPER COLUMBIA STEELHEAD ESU**. Organized by Major Population Groupings. Complexity categories: A = linear; B= dendritic; C= trellis pattern; D= core drainage plus adjacent but separate small tributaries.

Major Population Group	Population	Weighted Area Category	Complexity	
			Category	# MaSA (# MiSA)
Eastern Cascades	Wenatchee River	Intermediate	B	7 (8)
	Methow River	Intermediate	B	5 (5)
	Entiat River	Basic	A	1 (1)
	Okanogan River (US portion only) [1]	Intermediate	B	2 (6)
	Crab Creek (ext)	Intermediate	D	1 (2)

[1] Steelhead historically and currently occupy tributary habitat in both the U.S. and Canada. Current ICTRT analyses are focused on the US portion, although additional MSAs or populations may exist in the Canadian portion.

Snake River Fall Chinook and Sockeye Populations

The ICTRT adapted the approach for identifying major and minor spawning areas as follows to reflect biological characteristics of Snake River fall chinook and sockeye. Appendix B includes specific details of our analysis of the relative amount of historical spawning/rearing habitat within populations of these two ESUs.

The extant Snake River fall chinook population includes five MaSAs: the two mainstem reaches described above along with the lower reaches of the Clearwater, Grand Ronde and Tucannon Rivers. The lower reaches of the Imnaha and Salmon Rivers may have supported relatively low levels of fall chinook spawning and are considered part of the upper mainstem MaSA.

A number of lakes ranging widely in size within the Columbia River basin historically supported sockeye production (see appendix B). The relative productivity of sockeye populations is generally correlated with lake surface area (Burgner, 2001). With the exception of Redfish Lake, the Stanley Basin lakes have been at the lower end of the size range of the Columbia River basin sockeye lakes. Redfish Lake falls into an intermediate size category based on surface area.

We have little information on the within population structure of the Redfish Lake sockeye. Based on recent observations, sockeye spawn along the lake shore in October and November (Good et al., 2005). Given the extremely low levels of Snake River sockeye returns, initial recovery efforts are largely focused on improving survival rates of out-migrant smolts. More detailed information on the spatial structure of the Stanley Basin lake populations may be generated as recovery efforts progress.

Abundance and Productivity

Risk of extinction at the population level can be directly related to the combination of abundance and productivity of a particular population. The VSP guidelines for abundance and productivity developed by McElhany et al. 2000 provide the rationale for considering these two parameters in combination. The VSP guidelines for abundance recommend that a viable population should:

- Be large enough to have a high probability of surviving environmental variation observed in the past and expected in the future;
- Be resilient to environmental and anthropogenic disturbances; maintain genetic diversity; and support/provide ecosystem functions;
- Demonstrate sufficient productivity to support a net replacement rate of 1:1 or higher at abundance levels established as long-term targets;
- Demonstrate productivity rates at relatively low numbers of spawners that, on the average, are sufficiently greater than 1.0 to allow the population to rapidly return to abundance target levels after perturbations.

A viable population should exhibit an average abundance high enough to result in compensatory (density dependent) processes providing some resilience to annual perturbations. This resilience results from increases in relative productivity due to reduced density dependent effects when abundance fluctuates to lower levels (McElhany et al. 2000).

Marine survival is a major factor contributing to annual variability in return rates of Interior Columbia anadromous salmonid populations (e.g., Deriso et al. 2001, Zabel et al. 2006). Indices of marine survival for Interior ESUs demonstrate relatively high level of year to year correlation in annual returns. Achieving a desired risk level for populations subject to relatively high levels of autocorrelation in annual return rates may require a higher combination of abundance and productivity to provide for rebuilding from consecutive bad years (e.g., Morris & Doak, 2002).

ICTRT Abundance & Productivity Objective:

We developed the following objective for our population level abundance and productivity criteria considering the specific VSP guidelines summarized above:

Intrinsic productivity and natural origin abundance should be high enough that 1) declines to critically low levels would be unlikely assuming recent historical patterns of environmental variability; 2) compensatory processes provide resilience to the effects of short term perturbations; and 3) subpopulation structure is maintained (e.g., multiple spawning tributaries, spawning patches, life history patterns).

We developed a quantitative metric for evaluating the abundance and productivity of a population. Specifically, we defined "viability curves" (e.g., LCWTRT, 2003) for each ESU. A viability curve describes those combinations of abundance and productivity that yield a particular risk or extinction level at a given level of variation. The two parameters are linked relative to extinction risks associated with short-term environmental variability. Given a particular productivity level, populations at higher levels of abundance are more resilient in the face of year to year variability in overall survival rates than smaller populations. Populations with relatively high intrinsic productivity (expected ratio of spawners to their parent spawners at low levels of abundance) are also more robust at a given level of abundance relative to populations with lower intrinsic productivity.

Viability curves are generated via a population viability analysis (PVA) incorporating metrics representative of the target population. While PVAs can vary widely in levels of detail and quantification, all PVA applications include some means of assessing the risk of reaching a specified threshold or evaluating rates of change in abundance over time.

There is a general consensus among reviews of PVA applications on the importance of expressing the results of PVA analyses in an appropriate context, including explicit recognition of the potential influence of key uncertainties (e.g., Brook et al., 2002). Two broad categories of uncertainty can have a significant influence on the results of a PVA analysis: 1) uncertainty regarding the form of the relationship between parent abundance and subsequent production; and 2) uncertainties generated by limited abilities to include all potential environmental factors. We have explicitly recognized these factors in developing and presenting results from PVA models for Interior Columbia salmonid populations. We conducted sensitivity analyses relating model outputs to a range of values for key input variables. We contrast projected risk metrics under alternative mathematical forms of the underlying stock production relationship. We also simulated the potential influence of measurement errors on model input parameters and on projected risk levels. In addition, uncertainty regarding future environmental and human induced conditions that affect key population rates and processes should be taken into account in considering the implications of a PVA analysis. We incorporate alternative future environmental scenarios into our current status assessments.

Viability Curves: Key Components and Definitions

Generating a viability curve requires an estimated extinction or quasi-extinction threshold, an estimate of the variability in productivity, and a target risk level (e.g. 5% in 100 years). We describe the derivation of viability curves for application to Interior Columbia populations in Appendix A. A brief summary of our approach, including the rationale for particular input assumptions, is provided below.

A specific viability curve is defined as the combinations of abundance and productivity corresponding to a particular extinction risk (Figure 3). In general terms, high abundance combined with moderate productivity could provide the same extinction risk as that of a lower abundance but higher productivity. We incorporate a minimum abundance threshold into our viability curves to address genetic and spatial structure components of our general abundance and productivity objectives. Combinations of abundance and productivity falling above the curve would result in lower extinction risk, whereas points below the curve represent higher risk. We developed viability curves corresponding to a range of extinction risks (1%, 5%, and 25% level in 100 years). We use a quasi-extinction threshold to represent extinction in generating viability curves. We define our viability curves in terms of a simple linear Hockey-stick density-dependent relationship. A particular viability curve is a function of a set of representative assumptions regarding population dynamics and environmental variation. Sets of viability curves were generated using ESU-specific estimates of age structure and variability in brood year productivity (including autocorrelation in annual return rates). Theoretical studies have indicated that high autocorrelation in population abundance trend data can influence projected risks in a PVA analyses (e.g., Morris & Doaks, 2002, Wichmann et al. 2005). Our evaluations of Interior Columbia stream type chinook and steelhead data series indicated strong short term autocorrelation in abundance and productivity (see appendix A).

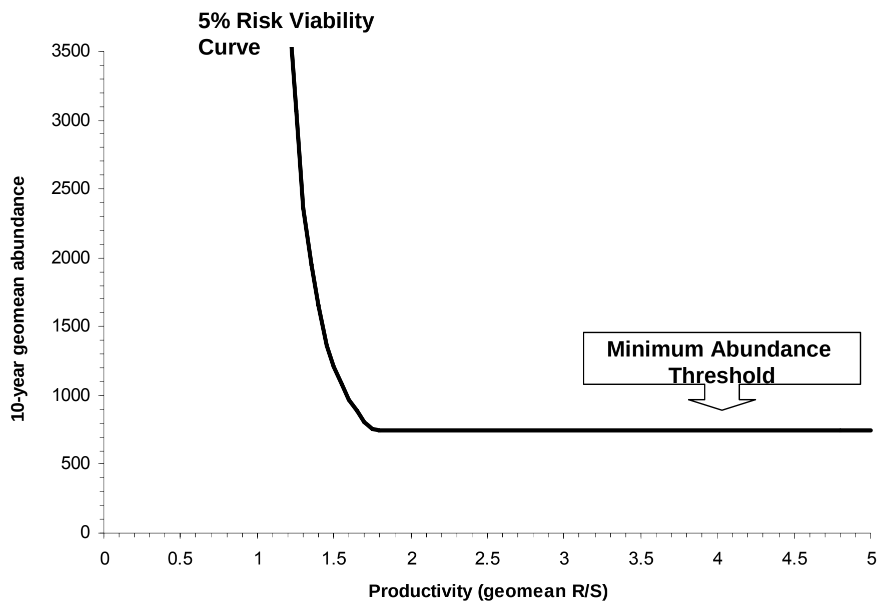


Figure 3. Example of a viability curve incorporating a minimum abundance threshold. The curve defines combinations of abundance and productivity values corresponding to a 5% risk of extinction over a 100 year period while maintaining average abundance at or above a minimum level set to avoid maladaptive genetic effects and to address spatial complexity objectives.

Risk Levels vs. Viable Status

The ICTRT population level viability criteria are expressed relative to an acceptable risk level of a 5% probability of extinction in a 100-year period. The level of risk is consistent with VSP guidelines and the conservation literature (McElhany et al. 2000; NRC, 1995). In addition, NOAA Fisheries has given previous policy guidance that a 5% risk of extinction over a 100-year period is an appropriate benchmark for population level risk assessment, at least for initial exploration. We chose to express the risk relative to a 100 year time from for several reasons; 1) it incorporates sensitivities to multiyear patterns/variations in environmental influences, 2) it is an appropriate time frame for considering recovery strategies that include habitat restoration actions that may take considerable time to result in survival improvements (e.g., restoring riparian habitat or stream structure to enhance parr to smolt survivals) and 3) a 100 year time frame subsumes short time frame risks. Under historical conditions, most populations within the region would have been rated as very low risk relative to the 5% viability curve. At the population level, recovery strategies should be targeted to achieving combinations of abundance and productivity above the 5% viability curve threshold. We recognize that alternative risk levels and time frames may be useful in assessing population status when considering short term effects of actions, etc.

Form of Spawner-Recruit relationship

We have provided ESU specific viability curves based on relatively simple and direct measures of abundance and productivity. In most cases, data used to evaluate current status will be based on a relatively limited number of years. Uncertainty levels and bias in parameter estimates can be very large. Therefore it is especially important that assessments employing fitted stock recruit curve parameters as an index of current productivity should directly incorporate considerations for sampling induced errors and bias in their assessments. We describe methods for minimizing the potential impact of sampling induced bias and error in the current status application section of this report.

We used a hockey-stick form of density-dependence to underlie our viability curves. However, we recognize that it is possible to express the productivity term in a viability curve in terms of a stock-recruitment function, e.g., Beverton-Holt or Ricker curves. There is substantial potential for error or systematic bias in estimates generated using curve fitting techniques, especially when a data series is relatively short and highly variable (e.g., Hilborn & Walters, 1992). Approaches to risk assessment based on empirical curve fitting should explicitly incorporate methods to reduce the impact of error and bias. In some cases, error or bias can be reduced by the choice of an appropriate statistical framework (e.g., Myers & Mertz, 1998, Mackinson et al. 1999, Michielsens & McAllister, 2004) or by incorporating independent variables that account for components of the overall variability in annual return rates (e.g., Morris & Doak, 2002).

Extinction Definition (Quasi-extinction thresholds)

We implemented a QET of 50 spawners per year over a consecutive four-year period in generating viability curves for application to Interior Columbia basin ESU populations. Four consecutive years represents a full brood cycle of adult (mature male and female spawners). A quasi-extinction threshold is defined as "..the minimum number of individuals (often females) below which the population is likely to be critically and immediately imperiled." (Morris & Doaks, 2002; Ginsburg et al. 1982). We selected 50 as a QET based on four considerations; consistency with theoretical analyses of increasing demographic risks at low abundance, uncertainty regarding low abundance productivity of Interior Columbia ESU populations due to the paucity of escapements less than 50 spawners in the historical record, sensitivity analyses indicating that the probability of multiple very low escapements increases substantially as the QET approaches 1 spawner per year, and consistency with applications by the Puget Sound and the Lower Columbia/Willamette TRTs (McElhany et al. 2003, 2006).

There is a substantial theoretical basis for employing a QET in population viability analyses (e.g., Morris and Doak, 2002). However, there is also a clear recognition of the problems inherent in identifying a single best fit value for any given population. It is generally recognized that relative productivity would be expected to drop off at extremely low abundance. Three factors contributing to highly elevated extinction risk at very low abundance are demographic stochasticity, Allee effects, and increased risk of permanently losing genetic variability. (e.g. McElhany et al. 2000). Demographic stochasticity reflects the impact of random events and processes at relatively small population sizes. Contributing factors would include mate selection, sex ratios and individual fecundity. Allee effects are reductions in relative productivity at low abundance due to factors such as ineffective mate pairing (Morris and Doak, 2002).

The Recovery Science Review Panel (RSRP) provided general guidance to the TRTs on the use of PVA models based on a literature review. The review supported the concept of a QET – recommending that "…PVA analyses be conducted evaluating the risk of population decline to a threshold N*, above which demographic stochasticity, Allee effects, and even genetic effects of inbreeding depression, can be largely ignored." The RSRP noted that demographic stochasticity generally can be ignored at mature population sizes of 100 and that more precise estimates for application in particular situations could be generated based on a ratio of estimated demographic to environmental stochasticities.

The productivity of Interior Columbia basin salmon and steelhead populations at very low annual spawning abundance is highly uncertain. We evaluated historical spawning abundance for Interior basin Chinook populations and found very few instances of spawning escapements below 50 until recent years (after 1985). The occurrence of annual spawning escapements below 50 is dramatically reduced if the data series is restricted to the pre-1975 period.

We carried out a series of sensitivity analyses relating QET levels to viability curve output parameters to probe the relationship between QET and projected extinction risk (Appendix A). While this analysis does not directly generate a specific number for use as a QET, it is clear from the frequency distributions of annual spawning levels that the proportion of years at low spawning abundance (below QET) increases rapidly as the numerical value of QET is adjusted downwards from 100.

The impact of repeated parent spawning years at such low levels on population productivity is a major uncertainty. This uncertainty contributed to our decision to maintain the QET in our population viability model runs at 50 spawners as a precautionary measure.

A QET value of 50 spawners per year for 4 years is consistent with values used in population viability analyses by the Puget Sound and the Lower Columbia/Willamette TRTs (Ruckelshaus et al. 2004, McElhany et al. 2006). The Puget Sound viability analyses (cited in app. D in McElhany et al. 2003) incorporate a QET value of "…62.5 spawners per year for four years.". The Lower Columbia/Willamette TRT initially used a QET of 50 in the viability analysis described in their initial draft viability report (McElhany et al., 2003). An updated version of their viability report includes an alternative viability modeling approach incorporating a QET that is a function of the relative size (amount of spawning habitat) of a population (McElhany et al. 2006). The new approach translates to a QET of 50 for smaller populations. For larger populations, the new approach would translate to a numerically higher QET, however McElhany et al. (2006) note that although it "[it] is tempting to conclude that since the new QETs are higher the criteria are more precautionary….the model used in 2003 (PCC) is different from the model in these benchmark curves, making direct comparison problematic." Oregon Coastal TRT incorporates the results from four different types of population viability models. Two alternative QET values are incorporated into each model, with the QET being expressed in terms that are consistent with the structure of the particular model (P. Lawson, pers. comm.). For example, the QET is expressed in terms of a minimum spawner per mile estimate a habitat based population model (Chilcote, 2005). In that particular application, the QET for a population will be a function of the minimum density estimate and the total miles of spawning habitat.

Reproductive Failure Threshold

The population viability models used by the ICTRT also incorporate a Reproductive Failure Threshold (RFT). While the QET is responsive to the number of spawners across a brood cycle, the RFT reflects uncertainty in the production from an extremely low return to the spawning grounds in a single year. If the number of spawners projected for a particular return year is at or below the RFT, production from that brood year is assumed to be 0. We have set the RFT for stream type chinook and steelhead populations to 10 spawners after reviewing updated run reconstruction results for Interior Basins Spring/Summer Chinook populations (Appendix A). Recent escapement levels are well below the documented historical ranges for these populations. Given the uncertainty and potential for increased demographic risks at relatively low population levels, we conducted three modeling exercises to inform the choice of an appropriate RFT value; an analysis of the potential for bias in estimating productivity at low parent spawning number, a simple demographic model of spawning success at low numbers, and an assessment of the relative risk associated with a 'wrong' choice RFT value (Appendix A).

The analysis of potential bias in estimating productivity as a function of spawning numbers indicated that bias in estimated returns from low escapement levels is likely for Interior Columbia data series, and that productivity estimates can be consistently inflated at low parent escapement levels, with the degree of bias increasing substantially for values below 20 spawners. The simple three spawning site model we developed to evaluate the potential impact of demographic effects at low spawner numbers indicated that the effective number of female spawners dropped off rapidly below 10 spawners.

Setting the Reproductive Failure Threshold (see below) at extremely low levels (e.g., less than 10 spawners in our sensitivity analyses) while maintaining the QET at 50 spawners per year over a brood cycle translates to a large increase in the expected proportion of spawning escapements below 50 fish across the 100 year projections. It is highly unlikely that these populations experienced such high proportions of very low spawning escapements historically.

Based on the results of these analyses and the observed returns at low escapements, we selected 10 spawners as a RFT for use in generating viability curves for yearling type chinook salmon and steelhead populations. We maintained the RFT at 50 spawners for Snake River Fall Chinook as a precautionary measure, recognizing the lack of data at very low spawning levels and the relatively large area that spawners can disperse over within the current population.

Minimum Thresholds for Abundance and Productivity

We have incorporated minimum thresholds for abundance into viability curves for application to Interior Columbia populations. Minimum abundance thresholds applied to the viability curves were based on the demographic and genetic rationale provided by McElhany et al. (2000) and reflect estimates of the relative amount of historical spawning and rearing habitat associated with each population. A minimum threshold value at or above 1.0 should also be applied to the population productivity parameter. Given a very high starting abundance, the relatively simple population model used to generate viability curves can, in some circumstances, project relatively low probabilities of extinction for average productivities below 1.0. In those cases the population would, by definition, be in long-term decline.

We incorporated a minimum abundance threshold of 500 spawners into the viability curves for populations in the Basic size category based on genetic and demographic considerations. Populations with fewer than 500 individuals are at higher risk for inbreeding depression and a variety of other genetic concerns (McElhany et al. 2000 and McClure et al. 2003 discuss this topic further). A minimum abundance of 500 spawners would appear adequate for compensatory processes to operate and to maintain within-population spatial structure for smaller Interior Columbia Basin salmon populations. However, for populations that cover big geographic areas with larger intrinsic potential, the ICTRT concluded higher minimum abundance levels were necessary to meet the full range of VSP criteria.

Incrementally higher spawning abundance thresholds were established for the remaining three population size categories (Table 3) We set thresholds for the two larger size categories (Large and Very Large) so that the expected average abundance at threshold levels was equivalent to approximately ½ of the density associated with achieving 500 spawners for a median sized population within the Basic category. Threshold levels for application to populations in the intermediate group were set so as to achieve median spawner densities at approximately half the range between the median population size for Basic and Large population groups. This density level represents a balance between using 500 as a minimum population abundance threshold regardless of the amount of spawning habitat and setting a population level threshold proportional to the amount of potential spawning habitat.). Increased thresholds for larger populations promote achieving the full range of abundance objectives including utilization of multiple spawning areas, avoiding problems associated with low population densities (e.g., Allee effects) and maintaining populations at levels where compensatory processes are functional. Setting the minimum abundance threshold in strict proportion to the estimated amount of potential spawning habitat implied unrealistic precision for each specific population and resulted in very high minimum abundance levels for larger populations.

Table 3. Minimum abundance thresholds by species and historical population size (spawning area) for extant Interior Columbia Basin stream type chinook and steelhead populations. Median weighted area and corresponding spawners per km (calculated as ratio with corresponding threshold) provided for populations in each size category (see appendix B).

Population Size Category	**Stream Type Chinook** (Upper Columbia Spr, Snake Spr/Sum ESUs)			**Steelhead** (Upper Columbia, Middle Columbia & Snake River ESUs)		
	Threshold	**Median Weighted Area (m X 10,000)**	**Spawners per KM (weighted)**	**Threshold**	**Median Weighted Area (m X 10,000)**	**Spawners per KM (weighted)**
Basic	***500***	23	21.7	***500***	141	3.4
Intermediate	***750***	44	17.1	***1,000***	382	2.6
Large	***1,000***	69	14.4	***1,500***	743	2.0
Very Large	***2,000***	145	13.8	***2,250***	1,175	1.9

Viability Curves for Interior Basin ESU Populations

We express our abundance and productivity criteria in terms of spawners. Measuring productivity and abundance at the spawning level reflects the cumulative impacts of all factors across the life cycle. The specific viability curves we provide in this report were generated using data from time periods of relatively constant harvest impacts. As a result, assessments based on comparing current spawner based abundance and productivity estimates to these curves effectively assume that recent average harvest rates will continue into the future. In some cases management or recovery strategies will include variable harvest rate strategies. The model we used to generate viability curves can be easily adapted to generate variations on the ESU specific viability curves that incorporate specific harvest rate rule sets.

We have generated specific viability curves for application to populations in each of the listed chinook, steelhead and sockeye ESUs in the Interior Columbia basin. We provide curves corresponding to risk levels of 25%, 5% and 1% over 100 years. Specific input values included age structure along with variance and autocorrelation estimates derived from time series of observed vs. expected brood year productivities (Appendix A). These values were averaged across populations within ESUs to generate representative viability curves (Table 4).

Table 4. Summary of average population input parameters used in generating viability curves for Interior Columbia Basin stream type chinook and steelhead ESUs. Variance and correlation estimates derived from time series of observed vs. expected brood year productivities.

ESU	Production Parameters				Average Age Composition			
	No. of trends	ESU Average Values ln (r/s)			3	4	5	6
		Variance	Adjusted Var.	Correlation Coefficient				
Snake River Sp/Sum Chinook	13	1.24	0.89	0.53	0.00	0.57	0.43	0.00
Upper Columbia Spring Chinook	3	0.95	0.51	0.68	0.00	0.60	0.40	0.00
Snake River Steelhead	6	0.39	0.25	0.60	0.03	0.60	0.35	0.02
Mid-Columbia Steelhead	12	0.40	0.18	0.74	0.03	0.46	0.43	0.08
Upper Columbia Steelhead [1]	18	0.38	0.20	0.69	0.02	0.38	0.45	0.15
Fall Chinook	1	0.45	0.25	0.67	0.53	0.43	0.04	0.00
Sockeye [2]	1	0.50	0.42	0.41	0.00	0.60	0.40	0.00

[1]Variance and correlation in natural return rates based on average for steelhead populations in Mid-Columbia and Snake River ESUs to avoid potential bias or masking effects of chronic high hatchery levels.
[2]Variance and autocorrelation for Wenatchee River sockeye used as surrogate for Snake River sockeye inputs.

The Willamette-Lower Columbia TRT has developed an alternative viability curved based method, the Population Change Criteria (PCC) approach (WL-LC TRT, 2003). This approach can be adapted to Interior Basin ESU viability curves for application to populations with relatively poor trend data sets.

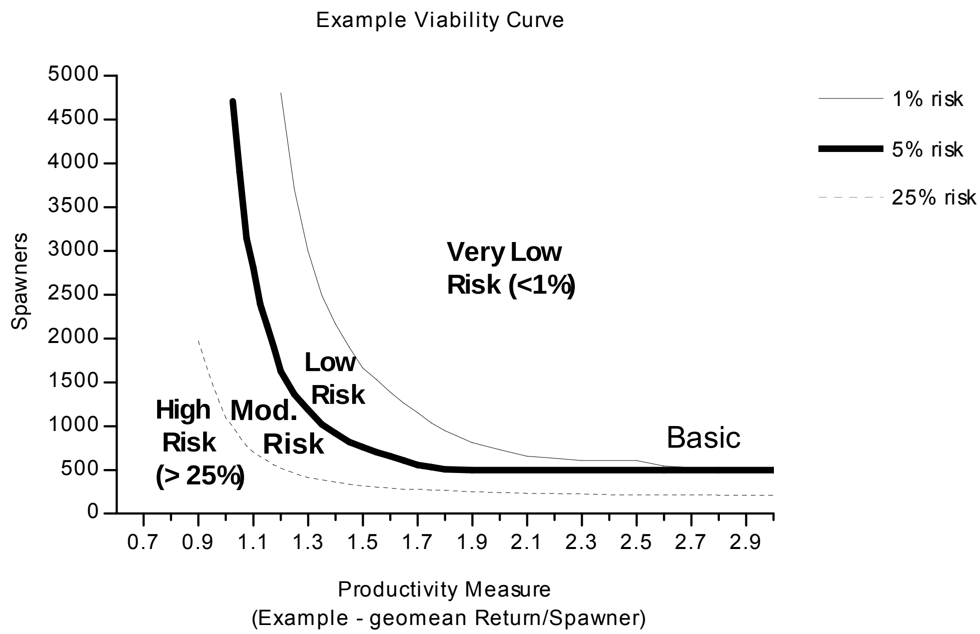
We encourage the development of metrics at other life stages, including juvenile productivity. Viability curves that incorporate specific measures reflecting survival from spawning to out migrating smolt and from out-migrant to adult return would address a major confounding factor, high year-to-year variability in marine survival rates. Incorporating smolt production measures would also aid in evaluating tributary habitat effects.

Stream Type Chinook and Steelhead

Viability curves were generated for use with two alternative productivity metrics: Return/Spawner and Annual population growth rate (lambda). The first is suitable for situations where detailed age structure and return data are available. Annual population growth rate (lambda) is provided as an alternative for use in situations where only index counts, or other types of counts without age structure are available. An example of a generic ESU viability curve in graphical format is provided in Figure 4. Graphic representations for all of the Interior Basin stream type Chinook and steelhead ESUs are included in Appendix A.

Figure 4a-b: Example of Viability Curves incorporating population size category threshold abundance levels.

a. Viability curve for application to populations in BASIC - small size category. Includes minimum average spawner threshold at 500.



b. Viability Curve including minimum population threshold of 1,000 spawners for use with Large- sized chinook populations.

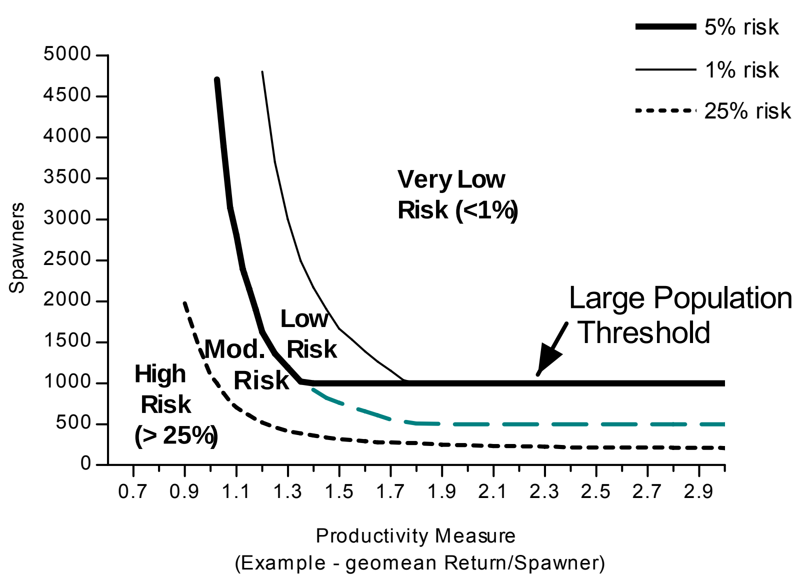


Table 5a. SNAKE RIVER SPRING/SUMMER CHINOOK. Population Viability curves in tabular format (return per spawner and population growth rate versions). Combinations of abundance and productivity exceeding these combinations would have a projected extinction risk of less than 5% in 100 years, assuming continuation of recent (1978-present) variation in return rates. Spawner/spawner estimates generated using Hockey-Stick recruitment function and average variance (0.89), autocorrelation (0.53) and age structure (0.57 age 4; 0.43 age 5) for populations in the ESU. Population growth rate based estimates generated using average running sums based variance (0.13) for ESU populations.

	Spawner to Spawner Measure				**Population Growth Rate (Lambda) Measure**				
	Minimum Abundance by Population Size Categories					**Minimum Abundance by Population Size Categories**			
Snake River Spr/Sum Chinook Growth Rate (S/S)	***Basic***	***Intermediate***	***Large***	***Very Large***	***Population Growth Rate***	***Basic***	***Intermediate***	***Large***	***Very Large***
1.15	5600	5600	5600	5600	**1.02**	27000	27000	27000	27000
1.175	4700	4700	4700	4700	**1.04**	8600	8600	8600	8600
1.2	3900	3900	3900	3900	**1.06**	4300	4300	4300	4300
1.25	3050	3050	3050	3050	**1.08**	2000	2000	2000	2000
1.3	2350	2350	2350	2350	**1.1**	2000	2000	2000	2000
1.35	1950	1950	1950	2000	**1.11**	1400	1400	1400	2000
1.4	1650	1650	1650	2000	**1.12**	1000	1000	1000	2000
1.45	1350	1350	1350	2000	**1.14**	880	880	1000	2000
1.5	1200	1200	1200	2000	**1.16**	630	750	1000	2000
1.55	1100	1100	1100	2000	**1.17**	560	750	1000	2000
1.6	970	970	1000	2000	**1.18**	500	750	1000	2000
1.65	890	890	1000	2000	**1.2**	500	750	1000	2000
1.7	810	810	1000	2000	**1.22**	500	750	1000	2000
1.75	760	760	1000	2000	**1.24**	500	750	1000	2000
1.8	720	750	1000	2000	**1.26**	500	750	1000	2000
1.9	650	750	1000	2000	**1.28**	500	750	1000	2000
2	600	750	1000	2000	**1.3**	500	750	1000	2000
2.1	550	750	1000	2000					
2.2	510	750	1000	2000					
2.3	500	750	1000	2000					
2.4	500	750	1000	2000					
2.5	500	750	1000	2000					
2.6	500	750	1000	2000					

Table 5b. UPPER COLUMBIA RIVER SPRING CHINOOK. Population Viability curves in tabular format (return per spawner and population growth rate versions). Combinations of abundance and productivity exceeding these combinations would have a projected extinction risk of less than 5% in 100 years, assuming continuation of recent (1978-present) variation in return rates. Spawner/Spawner estimates generated using Hockey-Stick recruitment function and average variance (0.51), autocorrelation (0.68) and age structure (0.60 age 4; 0.40 age 5) for populations in the ESU. Population growth rate based estimates generated using average running sums based variance (0.13) for ESU populations.

Upper Columbia Spring Chinook Growth Rate (S/S)	Spawner to Spawner Measure: Minimum Abundance by Population Size Categories				Population Growth Rate (Lambda) Measure	Minimum Abundance by Population Size Categories			
	Basic	*Intermediate*	*Large*	*Very Large*	*Population Growth Rate*	*Basic*	*Intermediate*	*Large*	*Very Large*
1.35	5400	5400	5400	5400	**1.02**	48000	48000	48000	48000
1.4	3800	3800	3800	3800	**1.04**	15400	15400	15400	15400
1.45	3100	3100	3100	3100	**1.06**	6600	6600	6600	6600
1.5	2700	2700	2700	2700	**1.08**	3950	3950	3950	3950
1.55	2400	2400	2400	2400	**1.1**	2300	2300	2300	2300
1.6	2100	2100	2100	2100	**1.104**	2000	2000	2000	2000
1.65	1850	1850	1850	2000	**1.12**	1400	1400	1400	2000
1.7	1600	1600	1600	2000	**1.14**	1050	1050	1050	2000
1.75	1400	1400	1400	2000	**1.145**	1000	1000	1000	2000
1.8	1300	1300	1300	2000	**1.16**	830	830	1000	2000
1.9	1100	1100	1100	2000	**1.18**	580	750	1000	2000
2	950	950	1000	2000	**1.2**	510	750	1000	2000
2.1	830	830	1000	2000	**1.21**	500	750	1000	2000
2.2	730	750	1000	2000	**1.22**	500	750	1000	2000
2.3	670	750	1000	2000	**1.24**	500	750	1000	2000
2.4	620	750	1000	2000	**1.26**	500	750	1000	2000
2.5	580	750	1000	2000	**1.28**	500	750	1000	2000
2.6	550	750	1000	2000	**1.3**	500	750	1000	2000
2.8	500	750	1000	2000					
3	500	750	1000	2000					
3.2	500	750	1000	2000					

Table 5c. UPPER COLUMBIA RIVER STEELHEAD. Population Viability curves in tabular format (return per spawner and population growth rate versions). Combinations of abundance and productivity exceeding these combinations would have a projected extinction risk of less than 5% in 100 years, assuming continuation of recent (1978-present) variation in return rates. Spawner/Spawner estimates generated using Hockey-Stick recruitment function and average variance (0.20), autocorrelation (0.69) and age structure (0.02 age 3; 0.38 age 4; 0.45 age 5; 0.15 age 6) for Interior Basin steelhead population trend data sets. Population growth rate based estimates generated using average running sums based variance (0.16) for ESU populations.

Upper Columbia Steelhead Growth Rate (S/S)	Spawner to Spawner Measure Minimum Abundance by Population Size Categories					Population Growth Rate (Lambda) Measure Minimum Abundance by Population Size Categories			
	Basic	*Intermediate*	*Large*	*Very Large*	*Population Growth Rate*	*Basic*	*Intermediate*	*Large*	*Very Large*
1	6600	6600	6600	6600	**1.02**	48000	48000	48000	48000
1.025	4700	4700	4700	4700	**1.04**	15400	15400	15400	15400
1.05	3800	3800	3800	3800	**1.06**	6600	6600	6600	6600
1.075	2850	2850	2850	2850	**1.08**	3950	3950	3950	3950
1.1	2150	2150	2150	2250	**1.1**	2300	2300	2300	2300
1.125	1800	1800	1800	2250	**1.104**	2000	2000	2000	2250
1.13	1650	1650	1650	2250	**1.12**	1400	1400	1500	2250
1.15	1450	1450	1500	2250	**1.14**	1050	1050	1500	2250
1.175	1200	1200	1500	2250	**1.145**	1000	1000	1500	2250
1.2	980	1000	1500	2250	**1.16**	830	1000	1500	2250
1.25	750	1000	1500	2250	**1.18**	580	1000	1500	2250
1.3	580	1000	1500	2250	**1.2**	510	1000	1500	2250
1.35	500	1000	1500	2250	**1.21**	500	1000	1500	2250
1.4	500	1000	1500	2250	**1.22**	500	1000	1500	2250
1.45	500	1000	1500	2250	**1.24**	500	1000	1500	2250
1.5	500	1000	1500	2250	**1.26**	500	1000	1500	2250
1.55	500	1000	1500	2250	**1.28**	500	1000	1500	2250
1.6	500	1000	1500	2250	**1.3**	500	1000	1500	2250
1.65	500	1000	1500	2250					
1.7	500	1000	1500	2250					
1.75	500	1000	1500	2250					

Table 5d. SNAKE RIVER STEELHEAD. Population Viability curves in tabular format (return per spawner and population growth rate versions).). Combinations of abundance and productivity exceeding these combinations would have a projected extinction risk of less than 5% in 100 years, assuming continuation of recent (1978-present) variation in return rates. Spawner/Spawner estimates generated using Hockey-Stick recruitment function and average variance (0.25), autocorrelation (0.60) and age structure (0.03 age 3; 0.60 age 4; 0.35 age 5; 0.02 age 6) for populations in the ESU. Population growth rate based estimates generated using average running sums based variance (.19) for ESU populations.

	Spawner to Spawner Measure					Population Growth Rate (Lambda) Measure			
Snake River Steelhead Growth Rate (S/S)	**Minimum Abundance by Population Size Categories**					**Minimum Abundance by Population Size Categories**			
	Basic	***Intermediate***	***Large***	***Very Large***	***Population Growth Rate***	***Basic***	***Intermediate***	***Large***	***Very Large***
1	4300	4300	4300	4300	**1.02**	27000	27000	27000	27000
1.025	3150	3150	3150	3150	**1.04**	8650	8650	8650	8650
1.05	2300	2300	2300	2300	**1.06**	4300	4300	4300	4300
1.075	1800	1800	1800	2250	**1.08**	2000	2000	2000	2250
1.1	1400	1400	1500	2250	**1.1**	1950	1950	1950	2250
1.125	1200	1200	1500	2250	**1.11**	1400	1400	1500	2250
1.13	1100	1100	1500	2250	**1.12**	1000	1000	1500	2250
1.15	940	1000	1500	2250	**1.14**	880	1000	1500	2250
1.175	830	1000	1500	2250	**1.16**	630	1000	1500	2250
1.2	720	1000	1500	2250	**1.17**	560	1000	1500	2250
1.25	550	1000	1500	2250	**1.18**	500	1000	1500	2250
1.3	500	1000	1500	2250	**1.2**	500	1000	1500	2250
1.35	500	1000	1500	2250	**1.22**	500	1000	1500	2250
1.4	500	1000	1500	2250	**1.24**	500	1000	1500	2250
1.45	500	1000	1500	2250	**1.26**	500	1000	1500	2250
1.5	500	1000	1500	2250	**1.28**	500	1000	1500	2250
1.55	500	1000	1500	2250	**1.3**	500	1000	1500	2250
1.6	500	1000	1500	2250					
1.65	500	1000	1500	2250					
1.7	500	1000	1500	2250					
1.75	500	1000	1500	2250					
1.8	500	1000	1500	2250					
1.9	500	1000	1500	2250					

Table 5e. MID-COLUMBIA RIVER STEELHEAD. Population Viability curves in tabular format (return per spawner and population growth rate versions).). Combinations of abundance and productivity exceeding these combinations would have a projected extinction risk of less than 5% in 100 years, assuming continuation of recent (1978-present) variation in return rates. Spawner to spawner estimates generated using Hockey-Stick recruitment function and average variance (0.18), autocorrelation (0.74) and age structure (0.03 age 3; 0.46 age 4; 0.43 age 5; 0.04 age 6) for populations in the ESU. Population growth rate based estimates generated using average running sums based variance (0.17) for ESU populations.

Middle Columbia Steelhead Growth Rate (S/S)	**Spawner to Spawner Measure** Minimum Abundance by Population Size Categories					**Population Growth Rate (Lambda) Measure** Minimum Abundance by Population Size Categories			
	Basic	*Intermediate*	*Large*	*Very Large*	*Population Growth Rate*	*Basic*	*Intermediate*	*Large*	*Very Large*
1.1	5650	5650	5650	5650	**1.02**	48000	48000	48000	48000
1.125	4200	4200	4200	4200	**1.04**	15400	15400	15400	15400
1.13	3900	3900	3900	3900	**1.06**	6600	6600	6600	6600
1.15	3300	3300	3300	3300	**1.08**	3950	3950	3950	3950
1.175	2500	2500	2500	2500	**1.1**	2300	2300	2300	2300
1.2	2050	2050	2050	2250	**1.104**	2000	2000	2000	2250
1.25	1550	1550	1550	2250	**1.12**	1400	1400	1500	2250
1.3	1200	1200	1500	2250	**1.14**	1050	1050	1500	2250
1.35	1000	1000	1500	2250	**1.145**	1000	1000	1500	2250
1.4	800	1000	1500	2250	**1.16**	830	1000	1500	2250
1.45	660	1000	1500	2250	**1.18**	580	1000	1500	2250
1.5	570	1000	1500	2250	**1.2**	510	1000	1500	2250
1.55	520	1000	1500	2250	**1.21**	500	1000	1500	2250
1.6	500	1000	1500	2250	**1.22**	500	1000	1500	2250
1.65	500	1000	1500	2250	**1.24**	500	1000	1500	2250
1.7	500	1000	1500	2250	**1.26**	500	1000	1500	2250
1.75	500	1000	1500	2250	**1.28**	500	1000	1500	2250
1.8	500	1000	1500	2250	**1.3**	500	1000	1500	2250
1.9	500	1000	1500	2250					
2	500	1000	1500	2250					
2.1	500	1000	1500	2250					

Snake River Fall Chinook

Snake River fall chinook exhibit important life history differences from stream type chinook and steelhead. Snake River fall chinook spawned primarily in large mainstem reaches and the dominant juvenile life history pattern was for subyearling migration. We calculated a viability curve for Snake River fall chinook (Figure 5) following the same analytical steps we applied to stream type chinook and steelhead ESUs.

We established a minimum abundance threshold for fall chinook consistent with the general abundance/productivity objectives summarized in the July 2003 ICTRT Viability draft report. We are recommending a minimum abundance threshold of 3,000 natural origin spawners for the extant Snake River fall chinook population. No fewer than 2,500 of those natural origin spawners should be distributed in mainstem Snake River habitat.

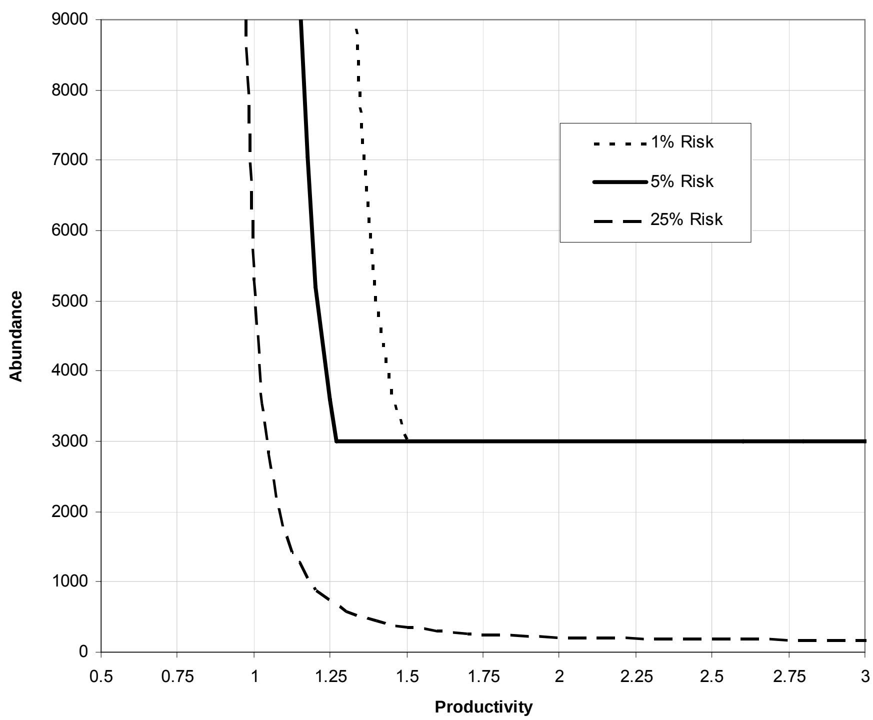


Figure 5. Viability curves for Snake River Fall chinook.

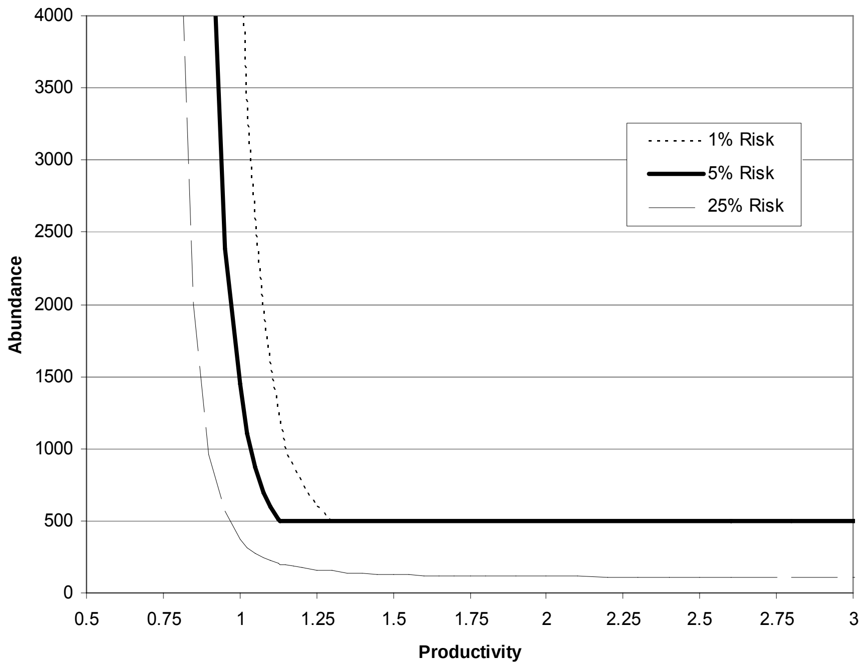
Snake River Sockeye

We generated two sets of curves for application to potential Stanley Lake Basin sockeye populations (Figures 6 a-b); these differ in their minimum abundance thresholds. More detailed descriptions of the relative size categories for Interior Columbia River Basin sockeye populations are provided in Appendix B. The Stanley Basin Lakes are relatively small compared to other lake systems that historically supported sockeye production in the Columbia Basin. Stanley

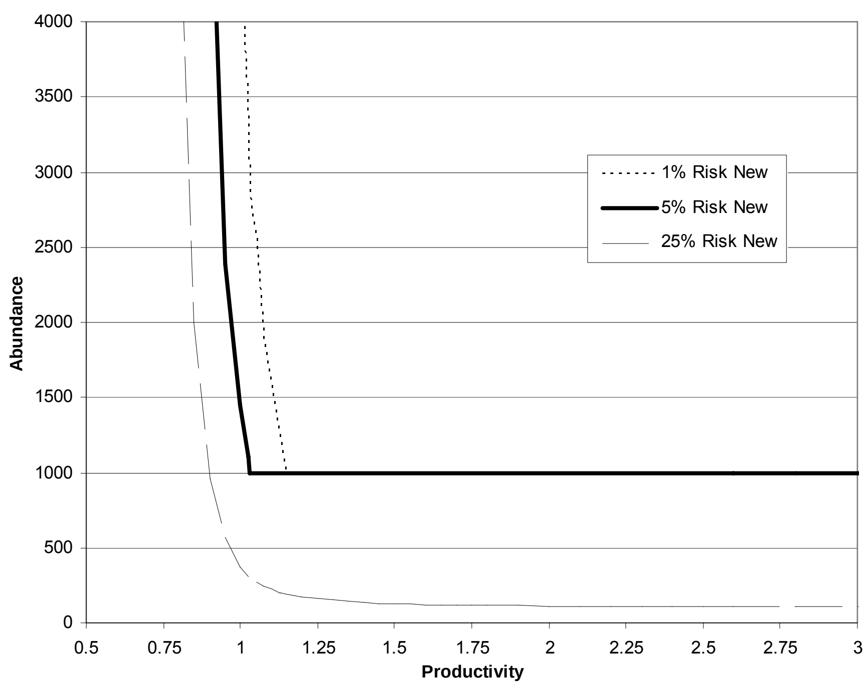
Lake is assigned to the smallest size category along with Pettit and Yellowbelly Lakes. Redfish Lake and Alturas Lake fall into the next size category – intermediate. We adapted the recovery abundance levels recommended by the Snake River Recovery Team (Bevan, et al. 1994) as minimum abundance thresholds. We set the minimum spawning abundance threshold at 1,000 for the Redfish and Alturas Lake populations (intermediate category), and at 500 for populations in the smallest historical size category (e.g., Stanley Lake). We used a run reconstruction of Lake Wenatchee sockeye as the basis for a representative set of variance and autocorrelation input values along with average age structure from historical Redfish Lake data (Appendix A).

Figure 6a-b. Viability curves for application to Snake River sockeye lake populations. A) Redfish Lake and Alturas Lake (Intermediate). B) small lake populations (Stanley Lake). Age structure used was 60% age 4 and 40% age 5 adult returns. Adjusted variance (variance unexplained by autocorrelation) and autocorrelation parameters (derived from Lake Wenatchee data) were 0.42 and 0.41, respectively.

a)



b)



Evaluating Population Status vs. Viability Curves

The underlying objective of the comparison of current status against a viability curve is to evaluate the relative likelihood that natural origin fish in the population of interest is capable of being self-sustaining. Comparing current status against the appropriate viability curve requires a measure of recent natural origin abundance and a measure of recent average intrinsic productivity. Intrinsic productivity is the expected production rate (expressed as a ratio of returns to spawn in future years vs. parent spawning numbers) experienced when spawner densities are low and compensation is not reducing productivity. The recent abundance metric must be measured in terms of spawners of natural origin. The measure of recent average productivity should reflect natural origin returns produced from the total number of fish directly contributing to spawning in the parental year. Hatchery origin natural spawners are counted as parents in the productivity calculations, and their natural origin offspring are counted as recruits and become natural origin parents in the next generation. In populations where a direct estimate of the relative productivity of hatchery origin spawners is available, the estimate of intrinsic productivity should be adjusted to reflect the rate associated with natural origin spawners.

Simple measures of current intrinsic productivity (both return/spawner and population growth rate metrics) can be influenced by the relative density of parent spawners. Most populations of listed Interior Columbia Basin stream type chinook and steelhead are currently at relatively low levels of abundance. As a result, adjustments to separate out the effects of carrying capacity are not necessary. However, as stock approach rebuilding target levels, direct estimates of intrinsic productivity can be affected by carrying capacity. There are options for addressing carrying capacity effects. Population growth rate approaches could employ threshold average spawning levels – if recent average total escapements exceed levels associated with carrying capacity effects, the expected population growth rate targets could be referenced to population maintenance (e.g., low likelihood average population growth rate is less than 1.0). Return per spawner series can be filtered, return per spawner pairs in which the parent escapements exceed a threshold associated with carrying capacity can be left out of the calculation of a recent average productivity.

The ICTRT has developed a relatively simple non-parametric approach for estimating productivity parameters for Interior Columbia salmon and steelhead populations. We describe and apply that approach in a separate report summarizing current status for Interior Columbia ESUs and their component populations (ICTRT in progress). In most cases, data used to evaluate current status will be based on a relatively limited number of years. Uncertainty levels and bias in parameter estimates can be very large. When sufficient data were available, we used a non-parametric approach to generate estimates of intrinsic productivity and the number of spawners associated with maximum production. We incorporated those estimates into a function in the form of a hockey stick recruitment function to generate viability curves.

We recognize that fitted stock recruit curves (e.g., Ricker, Hockey stick or Beverton Holt) are commonly used to estimate population productivity characteristics and as the basis for population viability analyses. There is substantial potential for error or systematic bias in estimates generated using curve fitting techniques, especially when a data series is relatively short and highly variable (Hilborn & Walters, 1992). Approaches to risk assessment based on empirical curve fitting should explicitly incorporate methods to reduce the impact of error and bias. In some cases, error or bias can be reduced by the choice of an appropriate statistical framework (e.g., Myers & Mertz, 1998, Mackinson et al. 1999, Michielsens & McAllister, 2004) or by incorporating independent variables that account for components of the overall variability in annual return rates (e.g., Morris & Doak, 2002).

Addressing Uncertainty in Assessing Current Status

Estimates of the current abundance and productivity of a population were based on sampling data and therefore were subject to some level of statistical uncertainty. The level of uncertainty, especially for the estimated productivity of a population, had a substantial impact relative to achieving targeted risk levels. The number of years included in the measures of recent abundance and productivity were a function of the specific methods used in generating measurements, the form of the criteria and the variance in annual return rates. Previous attempts to set recovery objectives (e.g., Bevan et al., 1995; Ford et al. 2001, McElhany et al., 2003) recommended minimum time series ranging in length from 8 to 20 years.

Sampling Induced Error

Preliminary sensitivity analyses indicate that directly incorporating a measure of the relative uncertainty in estimates of current productivity and abundance can reduce the potential for concluding that a population is at low risk when the 'true' risk level is actually high (type II error). Therefore, we recommend that current status estimates for comparison against the appropriate viability curve should include an adjustment based on the standard errors associated with point estimates of productivity and abundance. Preliminary evaluations indicate that the results are particularly sensitive to the estimate of intrinsic productivity.

We have evaluated three reasonable alternatives for buffering comparisons of current abundance and productivity for a population against the corresponding ICTRT risk metrics (Table 6). A more detailed explanation of these alternatives is included in Appendix A. We provide these examples as possible options to be considered in the recovery planning process, as well as to illustrate the relative sensitivity of status metrics to year to year variability and sampling uncertainties. Ultimately, choices regarding the treatment of uncertainty in decision-making include policy considerations.

Table 6. Alternative approaches for directly incorporating uncertainty into quantitative assessments of current status. Option A - simple probability based buffer, Option B1 two variations on a dual test approach designed to minimize the chance that the risk level being estimated is actually HIGH.

Option	Very Low Risk	Low Risk	Moderate Risk
A. Simple Probability Buffer	No less than **an 85% (approx. 1 stnd. error)** chance of being above the **1**% viability curve.	No less than **an 85% (approx. 1 stnd. error)** chance of being above the **5**% viability curve.	No less that a 50% probability of being above the **25%** viability curve
B.1 Dual Comparison: tolerance test to minimize chance that risk is actually High	No less that a 50% probability of being above the **1%** viability curve AND No more than a **1 in 100 (1%)** chance that the actual risk level exceeds **25%**	No less that a 50% probability of being above the **5%** viability curve AND No more than **a 1 in 20 (5%)** chance that the actual risk level exceeds **25%**	No less that a 50% probability of being above the **25%** viability curve
B.2 Dual Comparison: More precautionary tolerance test to minimize chance that risk is actually High	No less that a 50% probability of being above the **1%** viability curve AND No more than a **1 in 100 (1%)** chance that the actual risk level exceeds **10%**	No less that a 50% probability of being above the **5%** viability curve AND No more than **a 1 in 20 (5%)** chance that the actual risk level exceeds **10%**	No less that a 50% probability of being above the **25%** viability curve

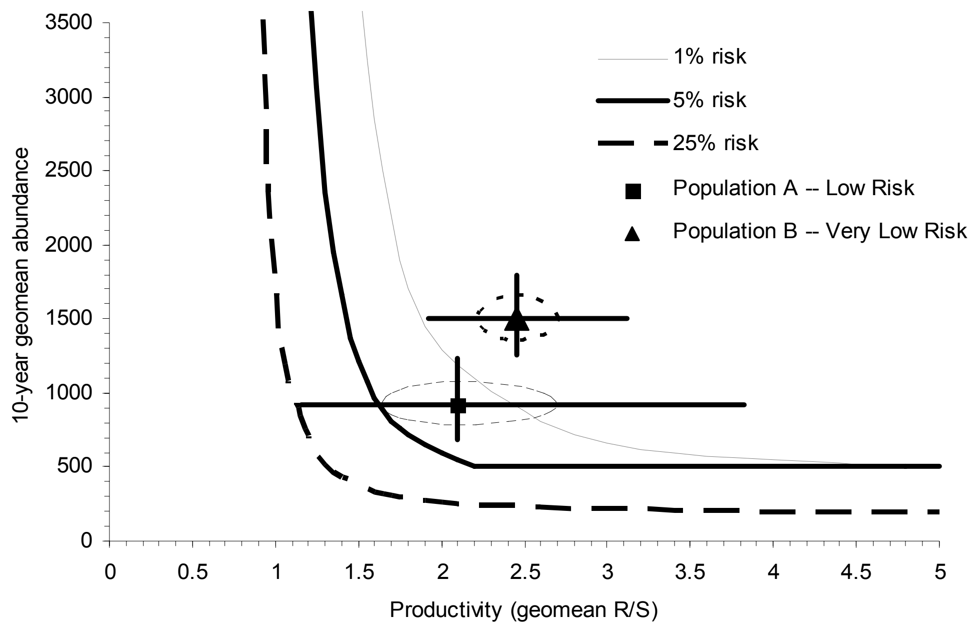


Figure 7. Evaluating the abundance and productivity of a population relative to the Viability Curve. Triangle and box symbols are point estimates of abundance and productivity for example populations. Ovals represent 1 standard error about mean values. Straight lines indicate 95% confidence limits on estimated abundance and productivity. Population A would be rated at Low Risk with respect to abundance/productivity, while Population B would get a Very Low Risk rating.

In general, all the analyzed options for treating uncertainty would result in a higher overall target (increasing the certainty that the population would “truly” be at or above the viability curve). Populations with higher variability require the greatest increases in the target, regardless of the option chosen. For a given variability level, the simple probabilistic buffer typically requires the greatest increase in the target, although there is some interaction with the level of variability of the population. Unlike the lower and moderately variable populations, a highly variable population would require a greater target to meet a stringent dual comparison than a simple probabilistic buffer. The following example illustrates the potential effect of using the alternative approaches for directly incorporating uncertainty associated with productivity estimates (Figure 8). The example is based upon the viability curves for a Very Large population within the Upper Columbia Spring Chinook ESU and includes a range of sample standard errors reflecting the levels calculated from recent data series for interior basin populations. These examples are based on an assumption that variation in mean productivity and abundance is multiplicative, following a lognormal distribution.

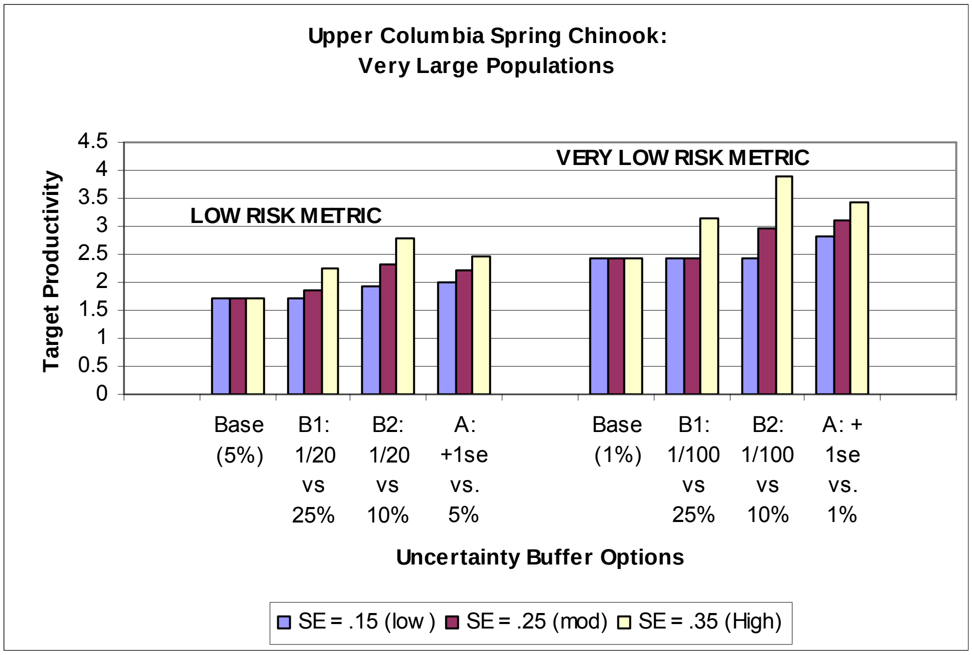


Figure 8. Example of the effects of alternative uncertainty buffers on the minimum productivity required at threshold abundance levels.

Spatial Structure and Diversity

Spatial structure concerns a population's geographic distribution and the processes that affect the distribution (McElhany et al. 2000). This distribution can affect population viability in several ways. For example, populations with a restricted distribution are more subject to loss due to a fine-scale environmental event (such as a single landslide) than populations with a more widespread or complex spatial structure (Isaak et al. 2003, Kallimanis et al. 2005). In addition, spatial structure can influence patterns of gene flow both within the population and between populations. It can thus affect a population's adaptation to local environmental conditions (Whiteley et al. 2004). Spatial structure's impact on extinction risk therefore spans both population dynamics and evolutionary processes (Morita and Yamamoto; Schrott et al. 2005).

Population-level diversity is similarly important for long-term persistence. Environments continually change due to natural process and anthropogenic influences. Populations exhibiting greater diversity are generally more resilient to these environmental changes in the short and long term. Phenotypic diversity, which includes variation in morphology and life history traits, allows more diverse populations to use a wider array of environments and protects populations against short-term temporal and spatial environment changes. Underlying genetic diversity provides the ability to survive long-term changes in the environment. Diversity criteria help ensure the preservation of the underlying genetic resources necessary for a population to fully exploit existing ecological opportunities, adapt to future environmental changes, or simply maintain a sustainable status. The emphasis must be on preservation, because once lost genetic variation is effectively gone forever (Riddell 1993). Riddell (1993) presented 10 principles for conserving diversity, primarily through the conservation of distinct reproductive groups. The focus of this strategy is to "manage Pacific salmon from the premise that localized spawning populations are genetically different, and valuable to the long term production of this resource." Populations and subpopulations (demes) were viewed as standard units for preserving diversity. The conservation of diversity could be achieved by "maximizing the spatial and temporal distribution of demes …maintaining populations with unique genetic traits or, genetic traits of importance, [or] maintaining populations occupying atypical habitats or expressing unusual phenotypic traits."

McElhany et al (2000) provide a number of additional guidelines for the spatial structure and diversity of viable salmonid populations that consider these principles. Specifically, their guidelines suggest that for spatial structure: a) habitat patches should not be destroyed faster than they are naturally created; b) natural rates of straying among subpopulations should not be substantially increased or decreased by human actions; c) some habitat patches should be maintained that appear to be suitable or marginally suitable, but currently contain no fish; and d) source subpopulations should be maintained. For diversity, they indicate that the important principles include: a) human-caused factors such as habitat changes, harvest pressures, artificial propagation, and exotic species introduction should not substantially alter variation in traits such as run timing, age structure, size, fecundity, morphology, behavior, and molecular genetic characteristics; b) natural processes of dispersal should be maintained. Human-caused factors should not substantially alter the rate of gene flow among populations; c) natural processes that cause ecological variation should be maintained. For both these parameters, a recommendation that uncertainty be accounted for in status evaluations is also included.

The ICTRT has used the general guidelines presented by McElhany et al. (2000) and Riddell (1993) to develop criteria with which to assess the robustness of a population. Because the spatial structure and diversity guidelines outlined are broadly overlapping (see above.), we consider these two parameters jointly. We consider all our criteria to be based on the conditions expressed by natural-origin fish. Finally, we follow the suggestion of McElhany et al. (2000) to use historical spatial structure and diversity as a default benchmark, since neither the precise role that diversity plays in salmonid population viability nor the relationship of spatial processes to viability are well-understood.

Interior Columbia Spatial Structure and Diversity Applications

Our viability criteria for spatial structure and diversity provide a measure of the status of a population. They address specific components of these parameters (or processes that affect these parameters), and thus also provide guidance for recovery actions to restore and/or preserve those populations. There is a good deal of uncertainty in many aspects of our spatial structure and diversity criteria, not least of which is due to the lack of well-developed theory about the impact of these parameters on population and meta-population viability. These criteria were developed to provide a consistent structure in which to consider spatial structure and diversity, even in those cases when expert judgment must be used. They are consistent with current understanding of these factors. As additional data and information become available, they may change – either in the values of the criteria associated with risk levels, or in the definition of the metrics themselves. If alternative approaches or data are available, they can and should be considered in a spatial structure and diversity assessment. However, the *intent* of our metrics is to identify those factors that have the potential to affect the long-term persistence of the population, and these principles should be preserved.

Structure of our Spatial Structure and Diversity Criteria

We express spatial structure and diversity viable salmonid population (VSP) guidelines in a hierarchical format that outlines the goals, mechanisms to achieve those goals, and examples of factors to be considered in assessing a population's risk level. We developed some examples of scenarios leading to various levels of risk. In this document, we use that structure to present metrics (along with examples) appropriate for assessing population status with respect to each mechanism, and ultimately with respect to our biological goals. For clarification, we present the following definitions:

G*oals* are the biological or ecological objectives that spatial structure and diversity criteria are intended to achieve. We have identified two primary goals:

1. <u>Maintaining natural rates and levels of spatially-mediated processes</u>. This goal serves to minimize the likelihood that populations will be lost due to local catastrophe, to maintain natural rates of recolonization within the population and between populations, and to maintain other population functions that depend on the spatial

arrangement of the population.

2. Maintaining natural patterns of variation. This goal serves to ensure that populations can withstand environmental variation in the short and long terms.

Mechanisms are biological or ecological processes that contribute to achieving those goals (e.g., gene flow patterns affect the distribution of genotypic and phenotypic variation in a population).

Factors are characteristics of a population or its environment that influence mechanisms (e.g., gaps in spawning distribution affect patterns of gene flow, which then affect patterns of genotypic and phenotypic variation). In some cases the same factor can affect more than one mechanism or goal. The distribution of spawning areas in a branched vs. a linear system, for example, can affect both patterns of gene flow *and* the patterns of spatially mediated processes, such as catastrophes.

Metrics are measured and assessed at regular intervals to determine whether a population has achieved goals, or to evaluate its current risk level. Each factor has one or more metrics associated with it.

Criteria are specific values of metrics that indicate different risk levels.

We summarize the association between our defined goals, mechanisms, factors and metrics in Table 7. When a factor affects more than one mechanism or goal, we listed it under the mechanism for which it is most directly relevant.

Table 7. Organization of goals, mechanisms, factors and metrics for spatial structure and diversity risk ratings.

Goal	Mechanism	Factor	Metrics
A. Allowing natural rates and levels of spatially-mediated processes.	1. Maintain natural distribution of spawning areas.	a. number and spatial arrangement of spawning areas.	Number of MaSAs, distribution of MaSAs, and quantity of habitat outside MaSAs.
		b. Spatial extent or range of population	Proportion of historical range occupied and presence/absence of spawners in MaSAs
		c. Increase or decrease gaps or continuities between spawning areas.	Change in occupancy of MaSAs that affects connectivity within the population.
B. Maintaining natural levels of variation.	1. Maintain natural patterns of phenotypic and genotypic expression.	a. Major life history strategies.	Distribution of major life history expression within a population
		b. Phenotypic variation.	Reduction in variability of traits, shift in mean value of trait, loss of traits.
		c. Genetic variation.	Analysis addressing within and between population genetic variation.
	2. Maintain natural patterns of gene flow.	a. Spawner composition.	(1) Proportion of natural spawners that are unnatural out-of ESU spawners.
			(2) Proportion of natural spawners that are unnatural out-of MPG spawners.
			(3) Proportion of hatchery origin natural spawners derived from a within MPG brood stock program, or within population (not best practices) program
			(4) Proportion of hatchery origin natural spawners derived from a local (within population) broodstock program using best management practices.
	3. Maintain occupancy in a natural variety of available habitat types.	a. Distribution of population across habitat types.	Change in occupancy across ecoregion types
	4.Maintain integrity of natural systems.	a. Selective change in natural processes or impacts.	Ongoing anthropogenic activities inducing selective mortality or habitat change within or out of population boundary

Distribution and Occupancy

Several of our metrics relevant for spatial structure and diversity are dependent upon a comparison between historical conditions or distribution and current distribution.

- *Historical or potential distribution.* We used our analysis of intrinsic potential (Appendix B) as our hypothesis of potential or historically-occupied areas. Specifically, we assumed that areas rated “high” or “moderate” in that analysis were occupied, for purposes of our spatial structure and diversity assessments.

- *Current distribution or occupancy*. Occupied areas are those in which two or more redds from natural origin spawners have been observed in all years of the most recent brood cycle (i.e. the most recent generation) and have been observed for at least half of the most recent three brood cycles (approximately 15 years for steelhead and chinook). A MiSA is regarded as occupied when it has two or more redds present over the previously defined time periods. A MaSA is regarded as occupied when it has two or more redds within BOTH the upper and lower half of the weighted spawning area within that MaSA over the previously defined time periods. Natural origin offspring of hatchery fish are included in current distribution or occupancy.

We recognize that data may not be available at the appropriate scale to thoroughly evaluate all populations against the range of metrics described below. For immediate needs, we assess current occupancy using agency-defined species distribution. Future monitoring should be structured to assess occupancy more rigorously.

Habitat that is currently accessible and suitable should not be considered occupied unless occupancy criteria are met within the habitat. We regard the current vs. historical distribution comparison to be critical for assessing population status, in which we determine which aspects of the population’s demographic and population-level characteristics put it at risk. However, we recognize that a comparison of areas that could be occupied to historical and current distribution is an important component of a limiting factors analysis, in which the aim is to determine the factors that need to be altered in the population’s environment to improve its status.

Addressing Uncertainty in Spatial Structure and Diversity Assessments

An assessment of spatial structure and diversity at the population level requires consideration of a range of factors and the certainty of the information used to assess risk.

Information certainty needs to be considered in the risk assignment for SS/D criteria. The confidence in the assigned risk level is directly related to the certainty in the data and information used to assess risk. We recommend a precautionary approach, raising the assigned risk to a higher level in circumstances where there is a high level of uncertainty inherent in the data or information available for a particular metric.

Uncertainties associated with the SS/D criteria (individually as well as in aggregate) can be classified into the following categories and subcategories:

A. Data quality for a particular metric for the population of interest

    a. Completeness of spatial and temporal coverage within a year

    b. Length of the time series of the metric

    c. Consideration of precision and accuracy for the metric

B. Surrogate information for a metric

    a. Information for a specific metric from a population deemed to have similar characteristics

    b. Using other information from surrogate metrics

C. No data or information available for a metric

There is considerable variation across ESUs, and among populations within ESUs, in terms of the particular categories and the relative level of potential uncertainty effects. Metrics for which there are no data(lowest level of certainty) are presently assigned a moderate level of risk. Risk levels for metrics for which the data are assigned high or moderate certainty should not be adjusted. When the certainty is low the risk rating should be increased by one level. We provide the following guidelines to aid addressing different levels of uncertainty that may be encountered in evaluating populations against specific SS/D metrics.

High level of certainty, for a specific metric, can be achieved when there is specific information for the population of interest and the data is spatially and temporally complete for each year in the time series. In addition, the time series must be of adequate length (see criteria and occupancy descriptions) and the data must have high level of precision and accuracy as it relates to the metric of interest.

Moderate level of certainty, for a specific metric, is assigned when there is at least surrogate information from a population deemed to have similar characteristics or surrogate metric information. The surrogate information should be spatially and temporally complete for each year in the time series, the time series must be of adequate length, and the data must have high level of precision and accuracy as it relates to the metric of interest.

An additional way of assigning a moderate level of certainty, for a specific metric, is when information for the population of interest does not meet the conditions described for the high level of certainty for one of the following characteristics: spatial and temporal completeness; time series length; or precision and accuracy.

Low level of certainty, for a specific metric, is assigned when surrogate information does not meet the conditions described for the high level of certainty for one of the following characteristics: spatial and temporal completeness; time series length; or precision and accuracy.

An additional way of assigning a low level of certainty, for a specific metric, is when information for the population of interest does not meet the conditions described for the high level of certainty for two or more of the following characteristics: spatial and temporal completeness; time series length; or precision and accuracy.

Spatial Structure and Diversity Criteria

Goal A: Allowing natural rates and levels of spatially-mediated processes

Spatially-mediated processes are those biological processes, such as gene flow, demographic exchange, local extirpation and recolonization that are influenced by the distribution and spatial organization of the population on the landscape. These processes are important both for mitigating risk of loss to local catastrophes and for maintaining normative levels of exchange among populations. We have developed an analysis of landscape intrinsic potential, or suitability for salmonid spawning (Appendix C); we use this analysis to characterize "natural" or "historical" distributions for this goal. If there is reason to believe that this hypothesis of distribution is in error, alternative historical distributions can be used, but the basis for those needs to be documented.

Mechanism A.1. Maintain natural distribution of spawning areas

We identified three factors that we consider under this mechanism:

1. Number and spatial arrangement of spawning areas

2. Current spatial range compared to historical spatial range

3. Change in gaps or continuities between spawning areas

Each of these factors addresses a different aspect of population distribution. The first addresses the inherent risk associated with different population configurations (e.g. linear vs. branched) in recognition that extinction risk is mitigated by physical separation of spawning habitats (Kallimanis et al. 2005). The second considers shrinkage or contraction of the distribution at its edges or extremes. These areas may be particularly important for maintaining connectivity with other populations (e.g. Dunham et al. 1997). The third factor considers changes of distribution within the population.

Factor A.1.a. Number and spatial arrangement of spawning areas

This metric addresses the inherent risk to the population owing to its natural configuration. Our criteria depend on the current number and arrangement of occupied MaSAs and other spawning habitat (Table 8). The dendritic pattern of rivers has been shown to have sometimes profound effects on extinction risk (Fagan 2002).

Table 8. Factor A.1a: Criteria describing risk levels associated with the number and spatial arrangement of occupied spawning areas.

Factor/metric	Pop. Group	Risk level			
		Very Low	Low	Moderate	High
Factor: Number and spatial arrangement of spawning areas Metric: Number of MaSAs, distribution of MaSAs, and quantity of habitat outside MaSAs	A,B,C,D	4 or more MaSAs in a non-linear configuration; or 3 MaSAs in a non-linear configuration plus one or more branches or MiSAs (outside of a MaSA) that sum to greater than 75% of the minimum habitat quantity of a MaSA (1 MaSA=100,000 m2 for spring/summer chinook salmon and 250,000 for steelhead)	2-3 MaSAs in a non-linear configuration separated by 1 or more confluences	2 or more MaSAs in linear configuration; or 1 MaSA plus one or more branches of MiSAs (outside of a MaSA) that sum to greater than 75% of the minimum habitat quantity of a MaSA or 1 MaSA with weighted intrinsic habitat quantity equal to or greater than the minimum needed for two MaSAs	≤ 1 MaSA

Factor A.1.b. Spatial extent or range of population

Reductions in the range of habitat used by a particular population can affect its vulnerability to local catastrophes. In addition, changes across significant habitat conditions (such as elevation) can affect life history or morphological diversity within a population (Frissell 1986). Finally, any change in range that increases or decreases the distance among populations may alter exchange of individuals between populations, hampering the exchange of genetic materials within an MPG and/or an ESU, and altering the likelihood of recolonization of extirpated areas (e.g. Bentzen Et al. 2001). We use occupancy of MaSAs across habitat conditions as our metric, reflecting the risk imposed by the current distribution of the population. (Table 9).

Table 9. Factor A.1.b. Criteria describing risk levels associated with spatial extent or range of population.

Factor/	Pop.	Risk Level			
Metrics	Group	Very Low	Low	Moderate	High
Factor: Spatial extent or range of population	A	Not attainable	All historical MaSAs occupied	50% or more of historical MaSAs occupied.	Less than 50% of historical MaSAs occupied.
Metrics: Occupancy of MaSAs across likely historical habitat conditions	B,C,D	Current spawning distribution mirrors historical (greater than 90% of historical MaSAs occupied)	Historical range reduced: 75% -90% of historical MaSAs occupied	Historical range reduced: 50%-74% of historical MaSAs occupied	Historical range reduced: less than 50% of historical MaSAs occupied

Factor A.1.c. Increase or decrease in gaps or continuities between spawning areas

Given the strong homing instincts of anadromous salmonids, significant changes in the distance between spawning areas may have impacts on gene flow within and among populations. The size of gaps between spawning areas may also affect the ability of a population to recolonize extirpated areas. A general dispersal distance relationship was used as one factor in defining distinct historical populations within Interior Basin ESUs (see ICTRT 2003 for further details). Based on that curve, dispersal or straying rates between spawning areas less than 10 km apart were relatively high. We suggest a simple index based on discontinuities between MaSAs (Table 10). The gaps criteria also incorporate consideration for the loss of spawning areas (MaSAs or MiSAs) at the lower end of populations. Such losses can substantially increase the distance from adjacent populations.

Table 10. Factor A.1.c. Criteria describing risk levels associated with a change in gaps or continuities between spawning areas.

Factor/ Metrics	Pop. Group	Risk Level			
		Very Low	Low	Moderate	High
Factor: Increase or decrease gaps or continuities between spawning areas	A,B,C,D	Population included 3 or more historical MaSAs AND All historical MaSAs currently occupied	75% or more of historical MaSAs occupied, gaps between MaSAs separated by 10 km or less	Currently occupied MaSAs separated by 10 km or more AND intervening historical spawning areas (MaSA or MiSAs) not occupied. OR Loss of MiSAs at lower end of population; increased distance to adjacent population by 25 km or more.	Occupied MaSAs separated by 15 km or more AND intervening historical spawning areas (MaSA or MiSAs) not occupied

Goal B: Maintaining natural levels of variation

This goal is aimed primarily at preserving existing genetic and phenotypic variation and, where natural patterns of variation have been altered, providing the conditions to allow that variation to be expressed. This variation or diversity is important for long-term resilience and adaptability. Relatively short-term (e.g., 5- to 10-year) observations of abundance and productivity alone are unlikely to be sufficient for the identification of a population's long-term risk of extinction because of inadequate diversity. Depending on the variability in environmental factors, many traits may not be expressed during the time intervals often used for assessing abundance and productivity. The establishment of diversity criteria provides the necessary mechanism for preserving a population's genetic resources during the recovery process, thereby increasing the likelihood of establishing or maintaining sustainable populations into the foreseeable future and beyond.

"Natural patterns and levels of variation" is not intended to specify a single point estimate of a trait (genetic or other), but rather the overall configuration of variation or potential that supported viable populations—encompassing range and distribution through time as well as average values. Thus, if a population historically occupied areas in which selective pressures alternated over long time periods (e.g. decades), the range of variation that allowed it to persist in that area should be preserved. Some judgment will be required in the application of the metrics supporting this goal, since historical patterns of variation are poorly, if at all, characterized. Potential sources of comparison for these metrics include historical information; other, more robust populations with similar characteristics; and expert judgment. These metrics provide a structure within which to consider variation, and outline the key elements that should be considered in any rating.

Importantly, in a relatively stable environment, a change in phenotypic mean away from a natural optimum can be considered as deleterious. However, Interior Columbia salmonids inhabit an environment that is not only changing now, but has also changed substantially over the last hundreds and thousands of years (e.g., Mantua and Hare 1994, Chatters et al. 1995). In addition, change in mean phenotype can also be indicative of a beneficial adaptive response of a population to an environment which has been altered, and for which a new natural optimum has been established. Two additional factors are thus important to consider while assessing populations with respect to this metric. The first is that not only the mean, but also the range of phenotypes or genotypes present in a population are important. An anthropogenic activity that maintains the same mean within a population, but dramatically reduces the variance should be considered selective, as the range of phenotypic expression has been dramatically reduced. In situations where the mean has changed in an apparently adaptive manner, care should be taken to ensure that the new "optimum" allows the population to be sustainable in other life stages and locations (e.g. genetic or environmental correlation between this trait and others should not reduce fitness at other life stages), and that a natural range of expression can still be achieved.

We identified four mechanisms that support our goal of maintaining natural levels of variation. We arranged these in a hierarchy, from direct measures of phenotypic and genotypic variation to indirect measures of environmental or other conditions that likely influence that variation. We include indirect measures for these two reasons. First, in many cases, direct measures of diversity are not available. Second, even when available, detectable change in phenotypic or genotypic measures may lag behind the impact causing that change. Including indirect, causal mechanisms thus serves to identify situations that are likely to become detectably impaired. Because the effect of these indirect measures on phenotypic and genotypic variation is in many cases less certain, we weight these indirect, causal mechanisms less heavily than direct measures.

Mechanism B.1: Maintain natural patterns of phenotypic and genotypic expression

This mechanism focuses directly on observed genotypic and phenotypic variation within populations and on changes in that variation. This is the variation that we seek to preserve in viable populations. Changes in these natural patterns are the strongest possible evidence that the population may be at risk with respect to diversity.

Factor B.1.a. Major life history strategies

Major life history patterns represent adaptations to environmental variation. We consider a major life history strategy to include a suite of phenotypic characteristics that are relatively correlated (at least phenotypically). Summer run-timing in stream-type chinook salmon, for example, rises to the level of a major life history strategy, as it encompasses not only adult run-timing, but also spawn-timing, age structure, size and to some extent, habitat preferences. Although life history strategies are a subset of phenotypic expression, we did not include this factor within "phenotypic variation" because we believe evidence indicates that these suites of characters were particularly important for overall population viability, and thus are less tolerant of loss or change in these characteristics.

Within an ESU, the dominant life history patterns may differ among populations in response to large scale patterns in environmental conditions or geographic patterns in habitat availability. Within a population, variations in life history patterns likely provide a buffer against high mortality in a particular year or habitat type (Healy, 1991). Particular combinations of adult run timing and spawning timing represent adaptations to the timing of flow and temperature conditions (Lichatowitch & Mobrand, 1995). Freshwater survival through juvenile rearing stages is an important determinant of overall productivity for stream type chinook and steelhead populations. A number of generalized movement patterns have been documented that could enhance survival through the summer and overwintering phases (ISAB, 1996; Reimers, 1973). Overwintering conditions in the relatively high elevation watersheds in the Interior Columbia can be extremely harsh. Late fall movements of a substantial proportion of age 0+ juvenile chinook and steelhead into downstream habitat areas afforded opportunities for increased survivals (Cramer et al. 2002). Loss or substantial reductions of a particular life history pattern could reduce the average productivity of a population.

We consider the following to be major life history strategies:

- Residence and anadromy
- Seasonal run-timing, including; spring- and summer- run in the Snake River spring/summer chinook ESU, winter and summer run steelhead, A and B-run steelhead
- Significant alternative juvenile migration patterns. These should include: consideration of timing of ocean migration (e.g. subyearling vs. yearling), relative distribution for summer rearing (e.g., natal tributary vs. downstream mainstem), and relative distribution for overwintering (e.g., natal tributary vs. fall downstream emigration

Our metrics for major life history patterns consider the presence and distribution of adult and juvenile life history strategies within a population (Table 11). In many cases, historical pathways will need to be inferred from habitat assessments and information from representative systems or from model based projections (e.g., EDT). In those cases key assumptions should be clearly described and justified.

Table 11. Factor B.1.a. Criteria describing risk levels associated with major life history strategies.

Factor	Pop. Group	Risk Level			
		Very Low	Low	Moderate	High
Factor: Major life history strategies Metric: Pattern (mean, range, etc.) of major life history expression within a population	A,B,C,D	No evidence of loss in variability or change in pattern	All historical pathways present, but some non-negligible change in pattern of variation	All historical pathways present, but significant (meaningful) change in pattern of variation	Permanent loss of major pathway (e.g. anadromy for *O. mykiss*, or loss of a juvenile pathway)

Factor B.1.b. Phenotypic variation

This factor includes morphological, life history, and behavioral traits. Because phenotypic traits are subject to natural and other selective events, the loss or severe truncation of specific traits reduces the resilience of a population to environmental perturbations, both in the short term (annual fluctuations, multiyear cycles) and long term (shifts in climatic conditions, etc.). We assess change in phenotypic variation by examining the mean, variation, and presence/absence of each trait (Table 12). Specific information on traits may not be available for all populations. Initial status reviews may be able to incorporate inferences based on information from similar populations within the same MPG or ESU. In addition, some case-by-case consideration may be necessary, due the range of conditions in the Interior Columbia. For instance, a population with an expanding range of spawn timing may be countering previous selective pressures that had truncated its range previously (a positive effect), or may be undergoing selection against the previous mean (a potentially negative effect). These types of considerations should be weighed in assigning a risk rating for this factor.

Table 12. Factor B.1.b. Criteria describing risk levels associated with change in phenotypic characteristics.

Factor/Metrics	Pop. Group	Risk Level: Very Low	Low	Moderate	High
Factor: Phenotypic variation Metric: Reduction in variability of traits, shift in mean value of trait, loss of traits.	A,B,C,D	No evidence of loss, reduced variability, or change in any trait	Evidence of change in pattern of variation in 1 trait (e.g., migration timing, age structure, size-at-age)	Loss of 1 trait or evidence of meaningful change in pattern of variation in 2 or more traits	Loss of 1 or more traits and evidence of change in pattern of variation in 2 or more traits; or change in pattern of variation of 3 or more traits (e.g., loss of a spawning peak and significant reduction in older age fish)

Factor B.1.c. Genetic variation

This factor addresses observed changes in genetic variation, regardless of the cause of that change (e.g., whether the change is due to introgression from non-local hatchery spawners or from the adverse genetic consequences of small population size).

We recommend that current and past population-specific genetic data sets be evaluated under four considerations:

- The amount of genetic variation detected within the population or subpopulations;
- The level of differentiation between subcomponents of the population
- The level of differentiation between the population and other populations (including hatchery stocks)
- Temporal change in levels of variation or differentiation within and between populations

These characteristics may be expressed by such measures as statistically significant reductions in heterozygosity, number of alleles, changes in allele frequencies, presence of non-native alleles, or as among locus (gametic) or within locus (genotypic) disequilibria consistent with ongoing or recent admixture with non-native populations.

However, we did not include specific genetic metrics or cutoffs in our criteria for three reasons. Most importantly, the wide variety of circumstances in the interior Columbia Basin requires a case-by-case examination of genetic data. For instance, available baseline genetic information may not be a reasonable picture of natural levels of genetic variation due to bottlenecks the population has experienced, or to extreme introgression from hatchery fish. Therefore, in some cases, change from a baseline might reduce the apparent risk to a population, whereas in others, the same degree of change might constitute a significant increase in risk level. Second, the ever-changing nature of molecular genetic techniques and analyses suggests that new advances may provide additional or improved methods to measure genetic variation. Finally, degree or magnitude of differentiation that could be gauged to be "high" or "low" will vary between species and data type and quality.

We do suggest risk levels associated with degree of change from "actual or presumed historical conditions" for genetic characteristics (Table 13). Requiring populations to show low levels of change from "actual or presumed historical conditions" is not meant to imply that the population must have the precise distribution of alleles that it had historically. Rather, we mean that the general pattern of differentiation exhibited within and between populations should be similar to that which existed historically (if a suitable baseline exists) or that which can be inferred as being likely from similar populations where reliable genetic inferences have been made.

Two issues relevant to categorizing a population with respect to this genetic criterion are worth particular mention. The first is the relatively slow response of neutral genetic markers to genetic drift. Populations that have been homogenized with each other, or with a hatchery stock, will not, if they maintain relatively large population sizes, show levels of differentiation consistent with those that existed historically in short time scales. In these situations, certain analyses can be used to assess whether the population merits a risk rating lower than is immediately apparent from its genetic characteristics:

– a fine-scale genetic analysis indicating that substructure within the population exists (i.e. that fish spawning in geographic proximity also show greater genetic affinity than they do to fish spawning more distantly). This structure should be confirmed across the population, and not be confined to a small portion. In addition, a sufficient number of generations to ensure high confidence in the results should be included;

– an analysis of genetic data indicating that the amount of divergence seen, even if differences between populations are not significant, is consistent with the time since the cessation of the perturbation and a very low level of exchange between populations. This analysis must include several samples both within and among the populations of interest;

– a robust analysis of patterns of dispersal. This would include sufficient spatial and temporal coverage to have high confidence that the population is neither receiving nor distributing out-of-population spawners at a rate that is above the expected frequency in natural situations, and that within population spawners are distributed in a manner consistent with natural situations. An analysis of this type is inferential with respect to our genetic criterion, and should thus be invoked with caution.

These analyses would be relevant for evaluating the characteristics of populations in the following management scenarios: re-introductions, re-building after population bottlenecks, and re-establishment of natural populations after an unnatural homogenizing event, such as overwhelming the population with hatchery-origin spawners.

Table 13. Factor B.1.c. Criteria describing risk levels associated with change in patterns of genetic variation.

Factor	Pop. Group	Risk Level: Very Low	Low	Moderate	High
Factor: Genetic variation Metric: Genetic analysis encompassing within and between population variation	A	No change from likely historical conditions	No change from likely historical conditions or evidence for a consistent trend towards historical conditions	Low level of change from likely historical conditions or evidence for a consistent trend towards historical conditions	Moderate or greater level of change from likely historical conditions
	B	No change from likely historical conditions	Low level of change from likely historical conditions or evidence for a consistent trend towards historical conditions	Moderate level of change from likely historical conditions or evidence for a trend towards historical conditions	Significant change from likely historical conditions
	C,D	No change from likely historical conditions	Criteria for A or B populations, dependent upon number of MaSAs in population	Criteria for A or B populations, dependent upon number of MaSAs in population	Criteria for A or B populations, dependent upon number of MaSAs in population

Mechanism B.2: Maintain natural patterns of gene flow

Maintaining natural patterns of gene flow is an indirect means of maintaining natural patterns of variation. We included spawner composition as an important factor supporting this mechanism. However, gaps within the population, and restrictions of spatial range (Factors A.1.b and A.1.c.) can also affect within and between population gene flow.

Factor B.2.a. Spawner composition

Natural breeding groups of Pacific salmon and trout (*Oncorhynchus* spp.) tend towards maintenance at natal localities because of strong homing capabilities coupled with localized adaptations (Hendry et al. 1998, 1999, NRC 1996, Reisenbichler et al. 2003). Stability of such aggregates over generations through centuries, and as fine as the local reach (Gharrett and Smoker 1993, Bentzen et al. 2001), is influenced by numbers of returning natal individuals (Waples 2004), ecological variability (Montgomery and Bolton 2003), and gene flow from exogenous fish (Utter 2001). This spatial and potentially adaptive level of variability within and between populations is recognized as important and necessary for viability of salmonid populations (McElhany et al. 2000).

The stability of salmonid population structure can be undermined by effective straying resulting from returning hatchery releases and natural-origin strays induced by anthropogenically altered conditions. Such increases of gene flow above natural levels are counterproductive to recovery efforts within listed ESUs because of hatchery adaptations or domestication (Epifanio et al. 2003, Waples and Drake 2004), losses of genetic variability through supportive breeding (Ryman and Laikre 1991, Wang and Ryman 2001), and erosions of natural population structure such as homogenization (Utter 2005). The ultimate impact of these increases in gene flow is dependent upon the duration of the increase, the proportion of exogenous spawners, and the origin of those spawners.

For this metric, we consider exogenous spawners to be all fish of hatchery-origin AND all natural-origin fish that are present due to unnatural, anthropogenically-induced conditions, but would not normally be present within the population. Upriver steelhead straying into the Deschutes River as an apparent result of unnatural high temperatures in the John Day reservoir would be one candidate for this category.

We have developed a flow-chart approach to assigning risk associated with exogenous spawners in salmonid populations (Figure 9). Our approach is sequential, and evaluators should consider exogenous spawners in their population in the sequence laid out. Our approach considers the source of the exogenous spawners first, providing increasing tolerance for both proportion and duration of exogenous spawners the more closely related they are to the population of interest. For exogenous spawners derived from the local population, we then consider the type of hatchery program from which those spawners were derived, allowing greater input from hatcheries using “best management practices.” Rather we suggest that hatchery programs that conform to the principles described in recent publications (e.g. Flagg et al. 2004, Olson et al. 2004, Mobrand et al. 2005) could be considered to have “best management practices.” These will change over time as our understanding of the impact of hatchery management practices on genetic, phenotypic and fitness characteristics increases. Main components of the program to be considered include brood stock selection, efforts to minimize within-population homogenization, actions to prevent domestication or other in-hatchery selection, breeding protocols, rearing and release protocols and other efforts to minimize effects on population structure and fitness components. Future assessments should consider advancements and updates in hatchery science when determining which category a particular program should be ascribed to.

These criteria are generally consistent with other efforts to quantify risk from hatchery origin spawners (Mobrand et al. 2005). However, we do encourage case-by-case treatment of conditions that may affect the risk experienced by the population. For instance, if exogenous spawners are localized within a large, complex population, leaving the bulk of the population unaffected, a somewhat higher proportion and/or duration of those exogenous spawners could be associated with a lower risk level. Similarly, in a very diverse MPG, the presence of exogenous spawners derived from a highly divergent population (even within that same MPG) might merit higher risk levels than shown. While we offer this flexibility, such situations should be well-documented and justified.

There are several more detailed considerations for applying our criteria. First, when assessing the current status of a population, conditions in the most recent three generations should be considered. Second, the proportion of spawners belonging to a category should be calculated using the total number of spawners in the denominator. Third, if there are multiple sources of exogenous spawners within a single population, the total proportion of exogenous spawners should be considered. In general, the highest risk level assigned to any of those sources should be used for this metric, unless there are two or more “moderate” rated sources, in which case a risk level of “high” should be used. However, there may be situations where spawners from each source would yield individually a low rating, but the total proportion of exogenous spawners is relatively high. In these cases, the risk rating should be increased appropriately to either moderate or high. Fourth, there may be cases where population specific estimates of the hatchery origin proportion of spawners are not available but circumstances indicate relatively high hatchery contribution rates are likely (e.g., nearby major release site, evidence for straying into other nearby natural areas). The risk rating applied in those cases should reflect the potential contribution levels of hatchery spawners. Finally, we do not extend our criteria beyond 5 generations for any source of exogenous spawners, because there is considerable uncertainty about the long-term impacts of this unnatural gene flow. We anticipate that future research will allow these criteria to consider longer time periods more robustly.

This metric offers the opportunity to contribute to planning efforts as well as to evaluate current risk. Conservation and/or supplementation programs may be desirable to mitigate short-term extinction risk, for example. In these cases, this metric provides a transparent means to plan and coordinate recovery efforts to minimize the risks from such a program.

Mechanism B.3: Maintain occupancy in a natural variety of available habitat types

Maintaining spawner occupancy in a natural variety of available habitat types is another mechanism to maintain natural patterns of variation. Differing habitats allow or promote the expression of differing phenotypes (Hendry and Quinn 1997, Hendry et al. 1998, Waples et al. 2001). Conceptually, the greater the range of habitat types available, the greater the potential for a population to express phenotypic diversity.

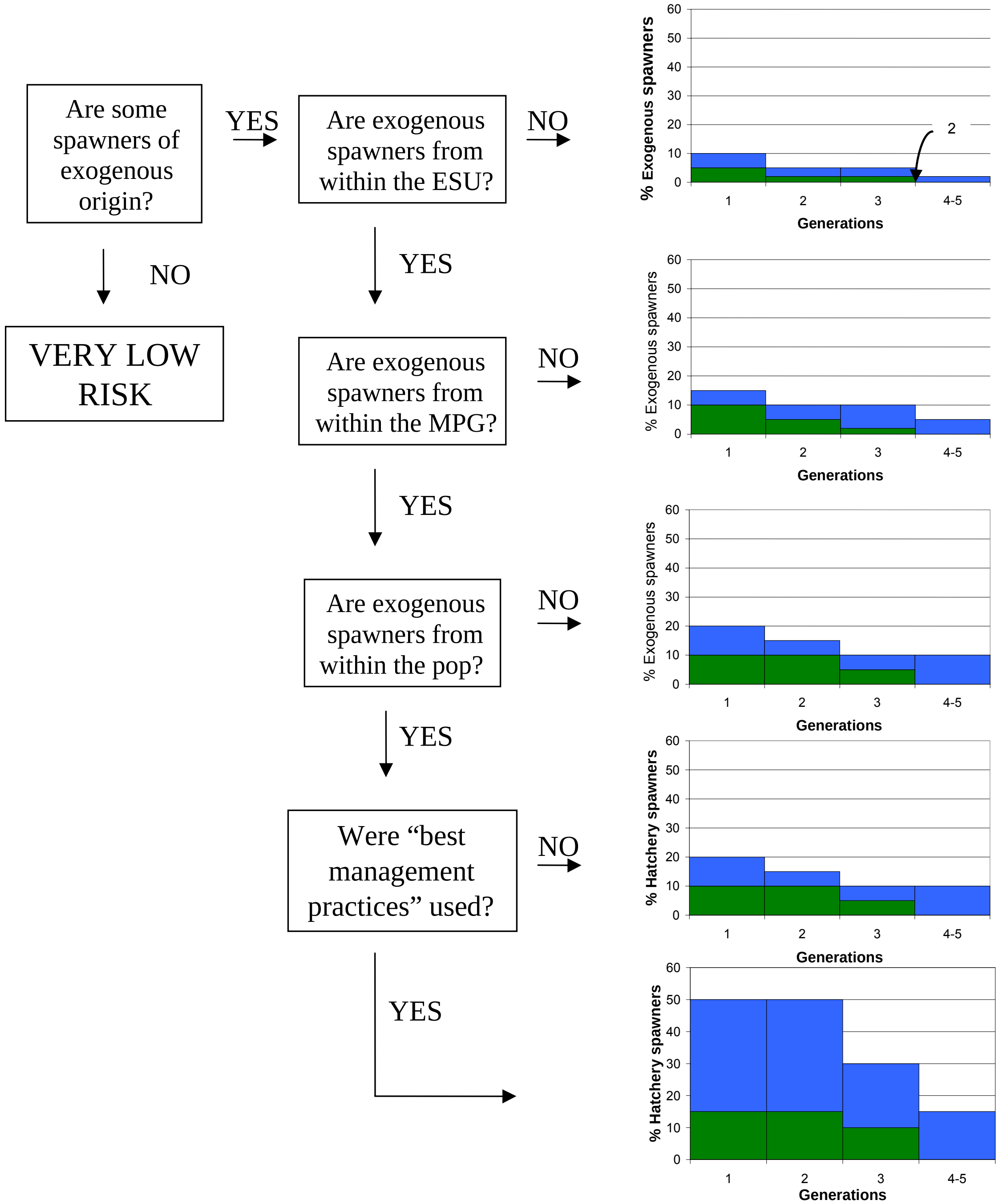


Figure 9. Risk criteria associated with spawner composition for viability assessment of exogenous spawners on maintaining natural patterns of gene flow. Green (darkest) areas indicate low risk combinations of duration and proportion of spawners, blue (intermediate areas indicate moderate risk areas and white areas and areas outside the graphed range indicate high risk. Exogenous fish are considered to be all fish hatchery origin, and non-normative strays of natural origin (see text).

Factor B.3.a. Distribution of population across habitat types

Salmonids regularly show local adaptations to habitat conditions they experience (Crossin et al. 2004). We rely on evidence that unique aquatic habitat types are produced within the context of the terrestrial ecosystems that encompass or border stream segments (e.g. Frissell et al. 1986). This relationship between a terrestrial ecosystem and its incorporated aquatic system is apt to be strongest for small streams and rivers and to be weaker for large rivers. We consider the range of habitat types occupied by a population as part of our spatial structure/diversity scoring system. A habitat diversity metric is intended to identify situations where that range of occupied habitats has changed substantively from its historic condition.

We use EPA's ecoregion classification (Level IV) (Omernik 1987, Gallant et al. 1989, Omernik 1995) to assess the historic (intrinsic) and current range of habitat types occupied. This was done by determining the distribution of intrinsic spawning habitat for a target population among the terrestrial ecosystems described by Omernik (1995). EPA Level IV ecoregion classification has the advantage of being widely accessible, well-documented and providing continuous coverage throughout the Columbia basin. These ecoregions were not developed with a focus on aquatic habitat, and their development variably includes attributes such as precipitation, land form, geology, and vegetation that influence aquatic habitat diversity. However, they are strongly correlated with differences in elevation, precipitation, and temperature regimes (ICTRT, unpublished data). Thus, as a first approximation, we believe that they capture reasonably some of the relatively substantive differences in habitat and environmental conditions that we are seeking to identify. We do note, however, that future work aimed at characterizing habitat diversity associated with population-level phenotypic and genetic diversity would be extremely useful for refining this metric. Among the likely tools for classification of habitat characteristics of biological relevance, we note some useful hydrological analyses, such as those developed by (Orsborn 1990, Lipscomb 1998).

Our approach to defining the relative risk associated with major shifts in distribution of spawners relative to ecoregions is illustrated in Figure 10. We define substantial changes in occupancy relative to historical distributions based on our intrinsic potential assessment. Ecoregions that supported more than 10% of the historical spawning area within a population are considered in the analysis. We defined a substantial change in relative distribution as a reduction of 67 percentage points or more in the relative distribution of spawning within an ecoregions that historically contained more than 10% of the weighted spawning area for a population. For example, if ecoregions X contained 50% of the total historical spawning area for a population, and that ecoregion currently represents 15% of the spawning area, the relative distribution has shifted by (50 - 15)/50 or 70%. In this case the shift would be counted as a substantial change.

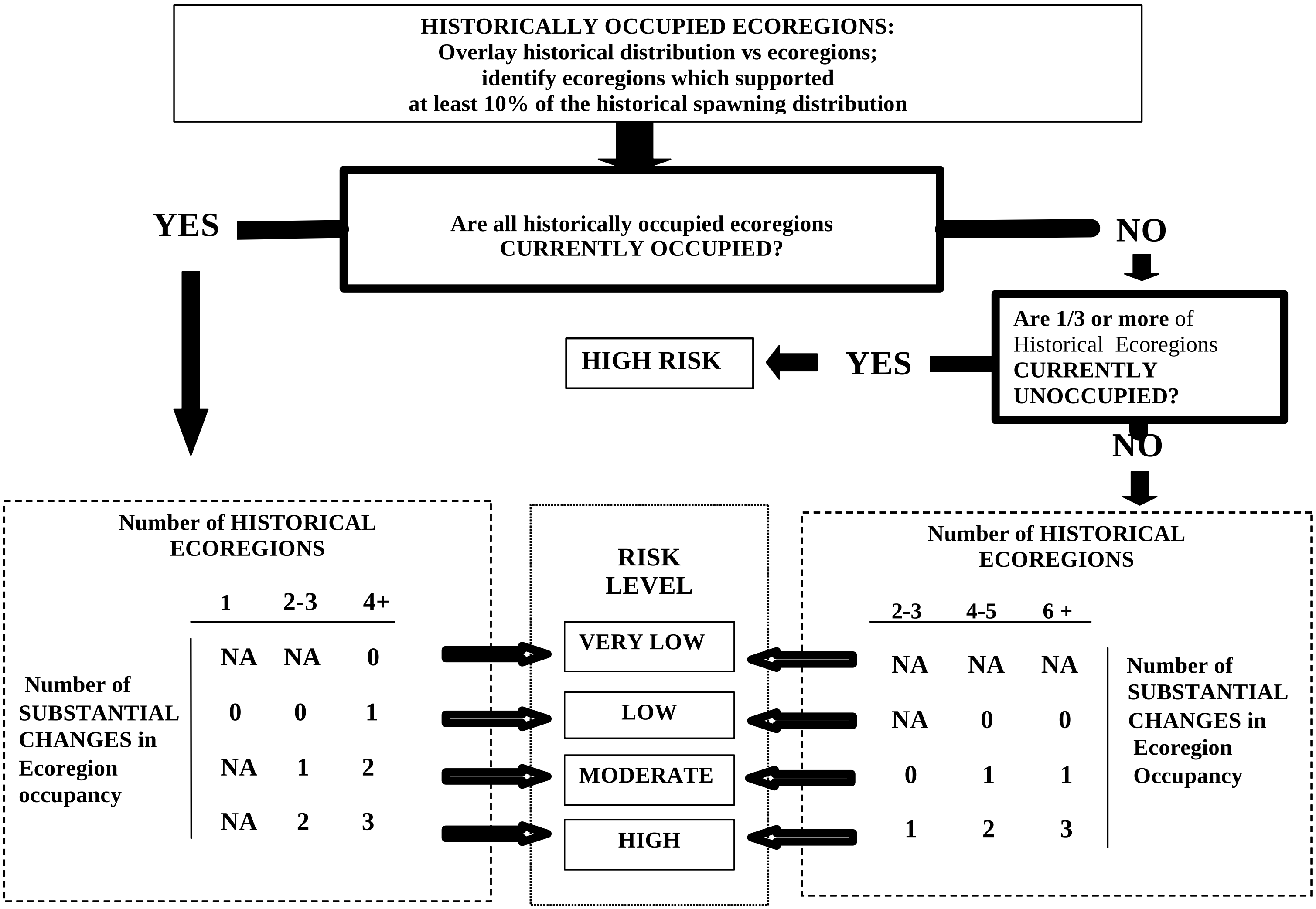


Figure 10. Evaluating changes in spawner distribution versus ecoregions.

Mechanism B.4. Maintain integrity of natural systems (Avoid selectivity in anthropogenic activities)

Maintaining the normative functioning of natural systems across the population's life cycle is an important component of maintaining natural patterns of diversity or variation. Disruption to the systems inhabited by natural salmonid populations can engender selective responses of these populations. For example, size-selective harvest has likely shifted size and/or life history traits (Handford et al. 1977, Ricker 1981, Healey 1986, Hamon et al. 2000, Hard 2004). Similarly, alterations to habitat conditions affecting the hydrograph, could substantially alter juvenile outmigration or spawn timing (Beechie et al., in press). Hatchery broodstock collection that preferentially removes one temporal component of a run could also have a selective impact on the natural population (McClean et al., Tipping and Busack 2004, 2003). Importantly, in identifying each of these activities it is not only that change in the system has occurred, but also that the change has a selective effect. In other words, that change causes a shift, truncation, or other alteration to the normal variation, and thus the fitness of the population, rather than merely a decrease in overall population survival or abundance, which is addressed in the abundance and productivity criteria. The selection may occur directly, through selective mortality or removal of individuals with a particular phenotype, or more indirectly, by reducing the fecundity or mating success of individuals with certain characteristics. Critically, the focus of this mechanism is on activities that affect normal variation rather than change in that variation itself (which is addressed in genotypic and phenotypic measures). The inclusion of this metric allows risks to diversity to be identified even in cases where phenotypic information is lacking.

Factor B.4.a. Change in natural processes or impacts

This metric aims to identify those activities that have the potential to cause substantial anthropogenic change in phenotypes in a relatively short time frame (e.g. 100 years). The magnitude of response to any selective force is determined by the heritability of a trait and the strength or intensity of selection (Lush 1937, Falconer 1960, Lynch and Walsh 1998). The "shape" or quality of that response is affected by the type of selection. In general, a force that selects for an optimum value will cause an exponential change in the value, ultimately reaching an asymptote, whereas a constant, directional force that selects against, for example, individuals at the largest end of the distribution, regardless of actual value will produce a more or less linear response (Figure 11). Of note, this linearity of response will only last as long as there remains sufficient genetic variation in the population to maintain a constant heritability. Eventually, persistent selection will deplete this variability, heritability will decrease in response, and the change in trait value will become asymptotic at a some physiologically limited maximum or minimum value (Hallerman 2003). Greater selection intensity will produce a greater response than a low intensity of selection (Figure 12). And, higher heritability will produce a more rapid and greater response (Figure 13) than will occur in a trait with a lower heritability.

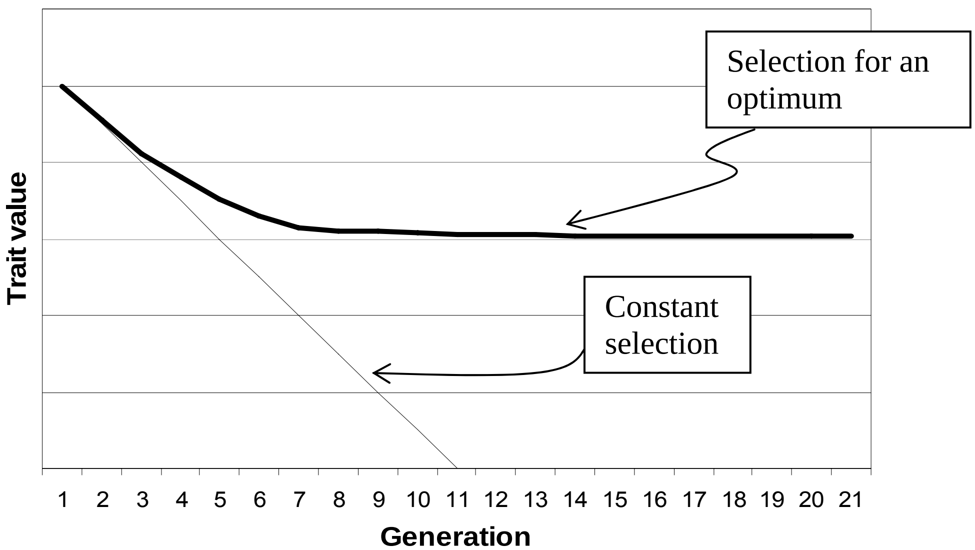


Figure 11. Constant, directional selection vs. selection for an optimum, given the same initial strength of selection and heritability. The y-axis in this graph is directionless, and is not intended to indicate that all selection will be against individuals with larger trait values. Asymptotic

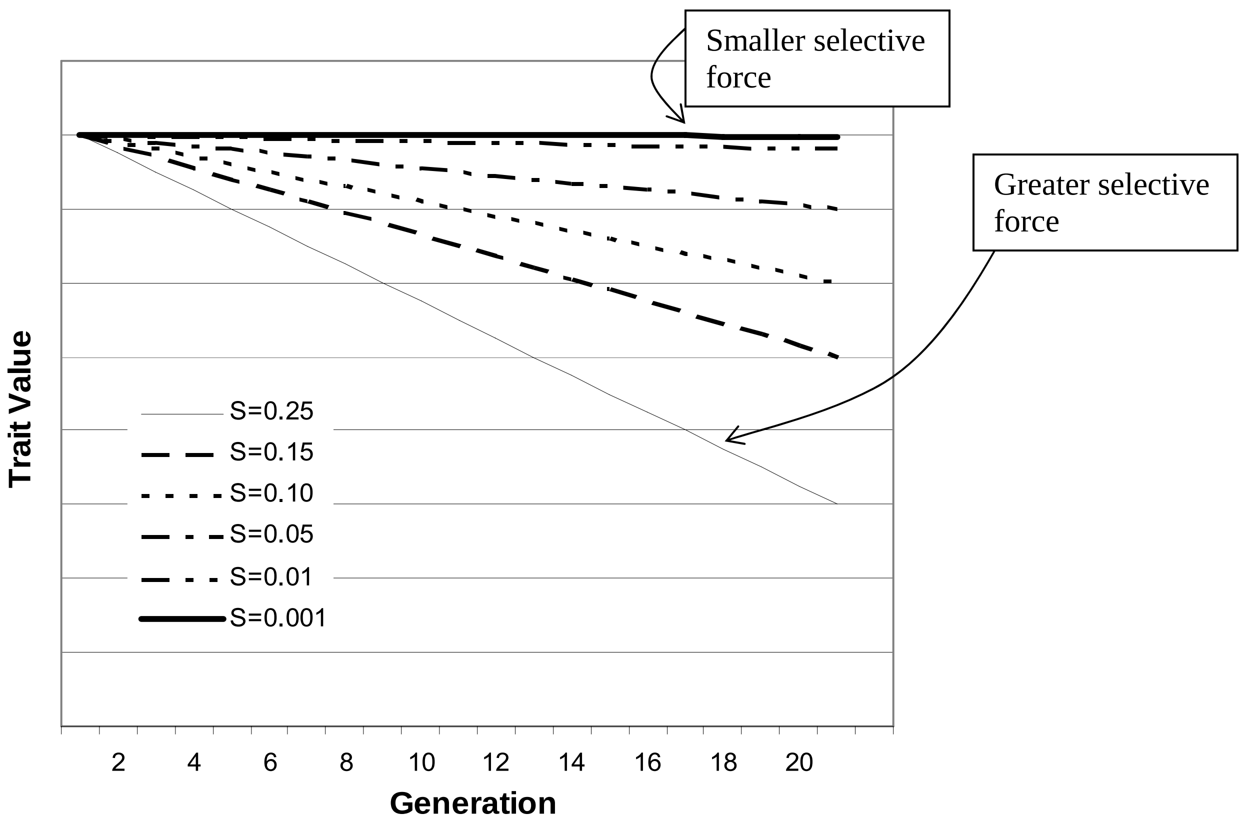


Figure 12. Trait response under varying strength or intensity of selection, with a constant heritability. The y-axis in this graph is directionless, and is not intended to indicate that all selection will be against individuals with larger trait values.

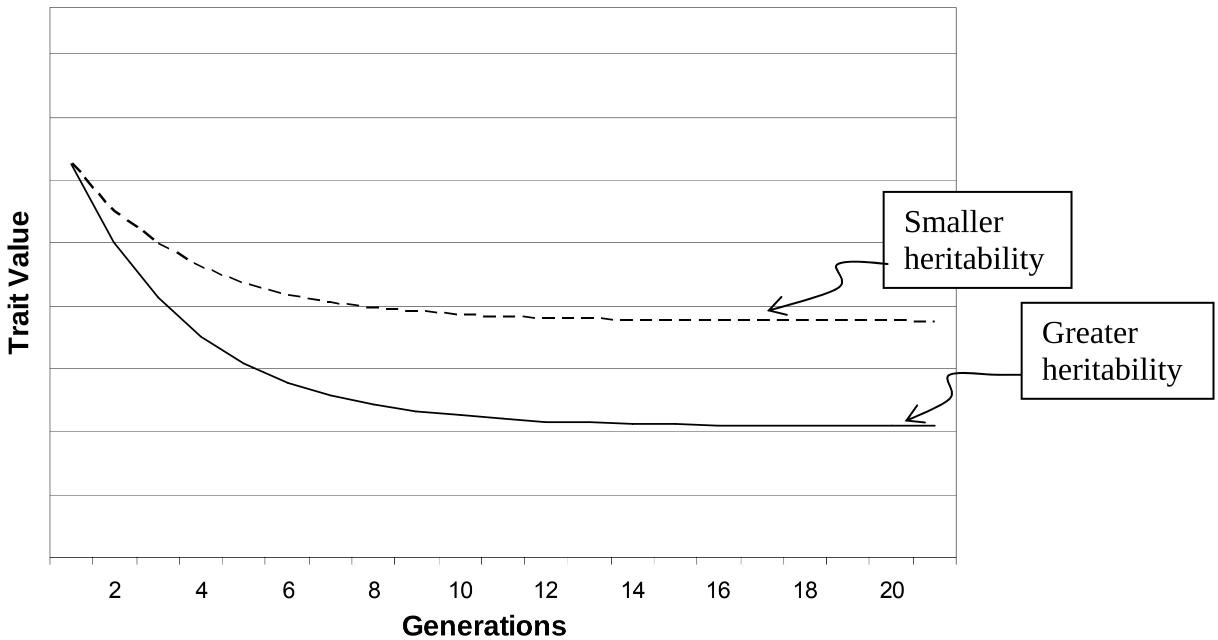


Figure 13. Differential response to selection for an optimum, under different heritabilities and the same selection intensity. The y-axis in this graph is directionless – the graph was drawn to show a decrease in the trait value; for other traits a decrease could be deleterious.

Assigning Risk Associated with Selective Activities

To assign risk associated with selective activities definitively, we would need to know the effect on mean fitness that the population change in phenotype (or in the range of phenotypes expressed) produced. Assessing these fitness effects, however, is very difficult, particularly for fitness traits in wild populations. Moreover, the phenotypes of poikilothermic animals, such as fish, appear to be more strongly developed in response to environmental influences, than homeothermic animals. In consequence, measures of heritability and of strength of selection for natural fish populations, are fairly limited in number and of poor precision (Hallerman 2003). In the absence of reliable quantified measures, we calculated the values of two phenotypic traits when reduced by standard proportions (Table 14) in order to assess qualitatively the potential ecological and demographic consequences of such a change. Given this information, we suggest that combinations of heritability and selection intensity that produce a 5-10% change in the mean should be regarded as moderate risk; combinations of heritability and selection intensity producing a change in the mean greater than 10% should be regarded as high risk. These suggested boundaries can be modified if additional information about these or other traits indicates that it would be appropriate.

Table 14. Proportional reductions in mean age at return (SRSS chinook) and length (SR fall chinook). This information was used to inform the suggested magnitude of change that would be associated with risk levels.

% Reduction	Mean Age at Return	Mean Length
0	4.2	85.0
1	4.2	84.2
2	4.1	83.3
5	4.0	80.8
10	3.8	76.5
20	3.4	68.0
25	3.2	63.8
50	2.1	42.5

In Figure 14 we present a decision process for assigning risk associated with selective activities. This framework considers the duration of the activity, the intensity of selection, and the heritability of the trait as factors that influence the magnitude of likely phenotypic response (Falconer 1960). Recognizing that empirical data describing the selection intensity on or heritability of a trait is very limited; we provide a qualitative illustration applying the metric in Box 1, and some general discussion below to assist with rating these factors.

Duration of the activity -- A selective activity that continues for less than a generation is much less likely to have a long-term effect on the population than one that has persisted for several generations. Those activities that have occurred for one generation or less can be regarded as very low risk. Intermittent activities (e.g. those felt in two out of every five years), which do not affect an entire generation, can also be regarded as lower risk than those that are continuous; however, intermittent activities that have occurred for protracted periods will have a larger effect. Finally, those selective activities that are ongoing are of greater risk than activities that have been discontinued, and we discount risk accordingly. Activities that have not occurred within five generations and are unlikely to be re-instated can be disregarded. (Note that the effects of these activities may still be perceptible in the population, but the point of this metric is to identify activities that are posing a risk to the population's diversity currently.)

Heritability – Heritability describes the proportion of phenotypic variation that is attributable to genetic variation, versus that which is environmentally determined. In general, morphological traits of organisms will tend to have relatively high heritability, while heritability for life history traits (presumed to have more direct association with fitness) will be low (Mousseau and Roff 1987, Falconer 1989). Nonetheless, maturation timing in several salmonid species has been shown to be among the most heritable of phenotypic traits in salmonids, with heritability values (h^2) of 0.50-0.65 (e.g. Dickerson et al. 2005, Kinnison et al. 1998, Hankin et al. 1993, Heath et al. 2002, Mousseau et al. 1998). Male body size, on the other hand, has been shown to be much more plastic in at least two species (Beacham and Murray 1988, Mousseau et al. 1998), with h^2 values of less than 0.3. However, it is unclear whether heritability measured in the laboratory is a good indicator of the heritability of a trait in the field (Weigensberger and Roff 1996, Hallerman 2003). Moreover, the heritability of a trait will be reduced through time as selection occurs (review in Hallerman 2003). Without a more complete understanding of trait heritability, there is no single cut-off value between “high” and “low” heritability categories for phenotypic traits, and a relative heritability should be considered. In general, those traits that are similar to spawning and migration timing in having some indication of a substantive genetic component can be considered to have “high” heritability; those that are substantially environmentally-driven at an individual level should be considered to have “low” heritability.

Strength of selection – Strength or intensity of selection will vary with the mean of the trait in the population before selection, the mean of the selected animals, the distribution of the trait, the proportion of the population affected and the type of selection (e.g. whether the selected animals are killed before they reproduce, vs. facing a small percent reduction in their fecundity). Actions that remove animals prior to reproduction will obviously have a greater selection intensity that those reducing fecundity slightly. Actions that change the difference between the means before and after selection will also have a higher selection intensity. Thus, situations that select against a relatively large component of the population at one end of the distribution, that select strongly against the likely natural mean, or that exert selection in a population with a relative narrow range of variation in the trait will all have higher selection intensities than the reverse situations. Actions that appear to affect relatively large components of the population or act strongly against the likely natural condition can be considered to have a “High” selection intensity. Actions affecting a very small component of the population can be considered to have a “Low” selection intensity.

Although this metric requires some application of judgment and review of previous work, we believe the value of identifying situations in which anthropogenic activities alter natural patterns of variation is high, particularly since so few populations have current or past phenotypic information available.

The TRT is reviewing the selective impacts of hydropower, harvest and hatchery activities affecting multiple populations within Interior Columbia ESUs that can be used in status assessments.

Assigning Risk in Populations Affected by Multiple Selective Activities

Some populations may be affected by more than one selective activity. Two issues are important for assigning risk in these situations. The first is identifying what component of the population has been affected. In cases where more than one activity affects the same component of a population (e.g. two activities both affect early out-migrants), those two activities should be treated jointly when working through the decision process outlined in figure 14. The second issue is devising a cumulative score for the multiple activities (or joint activities). In these cases, once all activities have been considered, each activity (or joint activities affecting the same component of the population) should be assigned a risk level using Figure 14. The population risk level is set at the highest risk level for any single factor in most cases. The single exception to this approach is the case in which three or more factors are all rated as moderate. In this case, we consider that the cumulative effect of those activities will likely be additive, and is sufficient to merit a high risk rating for the population.

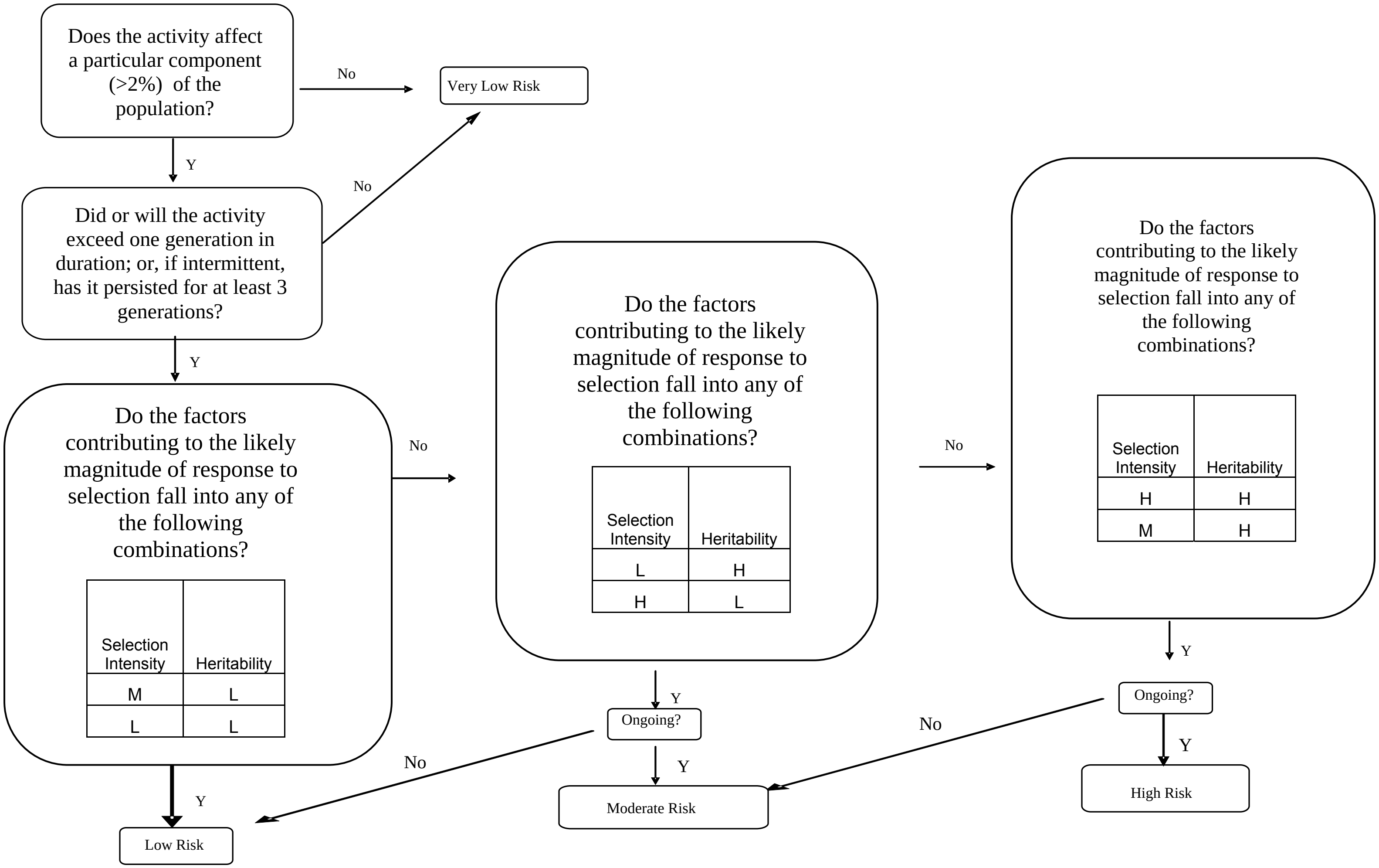


Figure 14. Decision process for assigning populations to a risk category associated with selective activities. Activities affecting the same component of the population should be considered simultaneously in this process. If multiple actions are selective in nature effect a single population that population will receive the highest risk category associated with a single action except in the case where 3 or more actions are associated with moderate risk in which case the population will be assigned to the high risk category for selective actions.

Box 1. Application of Selectivity Metric to a Hypothetical A-run steelhead population.

Scenario: In our hypothetical A-run steelhead population, 30% of the fish return as 1-ocean fish, and 70% return as 2-ocean fish. A fishery targeting this population removes approximately 8% of the returning 1-ocean fish, but 14% of the returning 2-ocean fish, because they are larger. Thus, 2-ocean fish are slightly, but disproportionately affected on two traits: their age of maturation and their length.

Change in the mean: Without the fishery, the average age of returning fish is 1.7 years:

Mean age = sum of (proportion of fish at each age*age)

Mean age = (0.3*1)+(0.7*2)=1.7

With the fishery, the mean of the fish left to reproduce is slightly different. One-ocean fish make up 71.4 percent of the population, 2-ocean fish make up 28.6 percent and the mean changes accordingly:

Mean age = (0.314*1)+(0.686*2)=1.686

Interpreting the change in mean: This change in mean yields a difference, or selection differential of 0.014 (see Figure 2 for the effect of alternate selection differentials on expected magnitude of response). If the heritability of the trait in question is high, as is age of maturation, this metric would receive a “moderate” rating. [If the proportion of 2-ocean fish had been lower (e.g. 30%), the total proportion of the population affected would have been less than 5% and the rating would be very low.] The same process could be followed for length, but the heritability would be low. These ratings would be decreased if the action is no longer ongoing.

Generating a Final Spatial Structure and Diversity Rating

Table 15 provides the "tool" or framework to integrate these several metrics and determine a population's composite risk level associated with spatial structure and diversity (SS/D). The table is organized hierarchically with the two primary goals of the SS/D criteria (McElhany et al. 2000) in the leftmost column. For each goal, one or more mechanism to achieve that goal is given in the next column. In general, these mechanisms describe the conditions associated with natural healthy populations. The third column lists the factors associated with each mechanism. Factors in this context are individual and population-level attributes that characterize each mechanism. The metrics outlined in the fourth column are the quantitative and qualitative measures used to assess a population's risk status relative to each metric.

The next four columns are a mirror image of the first four and provide the rules under which each metric score is assimilated up the hierarchy of the risk table. Risks are entered at the metric level and then carried through to higher levels to the right. For example, at the Factor level, metric A.1.a is assigned the risk level it was given at the Metric level. For comparison, B.2.a metrics 1-3 are integrated at the factor level following the rule set provided in the table. Metric scores across the entire table are integrated in a similar manner until the final column is reached which provides the population-level risk associated with SS/D.

The rules governing the integration at each level are intended to reflect the effect each metric would have on SS/D. Factors expressed in terms of direct metrics are integrated at the mechanism level by calculating the mean of the three metrics, effectively assigning a higher weight to direct measures of SS/D criteria. At the goal level the mean of the direct metrics is used for the same reasons. In those cases where the mean ends in a decimal part of 0.5 or less, round down to the higher risk level. The lowest score (highest risk) from the three B1 metrics is carried through the table to the factor and mechanism levels. To the extent possible, B1 metrics are measured deviations from natural patterns of phenotypic or genotypic expression. Thus, any measured deviation is likely to be an indicator of undetected changes and constitutes a substantial risk at the SS/D level. These are direct measures of phenotypic or genetic change in the population, and are given the highest weight in the overall integration of the B metrics. B2 metrics describe the influence that hatchery stocking may have on natural patterns of gene flow. In general, these metrics are integrated in the same manner as B1 metrics, the highest risk is carried through to the factor and mechanism levels. However, the case in which two or more of the metrics are rated moderate provides two complementary lines of evidence that hatchery stocking is altering the natural conditions and the risk level is increased to high accordingly. Factors B3 and B4 have a single metric the score of which is carried to the factor and mechanism levels. The B-type metrics are integrated at the goal level either by taking the B1 mechanism score or by using the mean of mechanism scores B.1 – B.4, whichever yields higher risk. This approach recognizes that B1 mechanisms are direct measures of deviations from natural conditions and should be given increased attention over the remaining B metrics. The overall population risk level is determined by using either the A-goal or B-goal score, whichever is lower (highest risk).

Table 15. Scoring system for deriving a composite, population-level spatial structure and diversity risk rating. Metrics and descriptions in the "Assessed Risk" column indicate contribution of individual metrics to integrated population score (Scoring: Very Low = 2, Low = 1, Moderate = 0, High = -1).

Goal: Mechanism	Factor	Metrics	Assessed Risk			
			Factor	Mechanism	Goal	Population
Goal A: 1. Maintain natural distribution of spawning areas.	***a. number and spatial arrangement of spawning areas.***	Number of MaSAs, distribution of MaSAs, and quantity of habitat outside MaSAs.	A.1.a	Mean of A.1.a., A.1.b, A.1.c.	Mean of A.1.a., A.1.b, A.1.c.	Lowest score (highest risk)
	b. Spatial extent or range of population	Proportion of historical range occupied and presence/absence of spawners in MaSAs	A.1.b			
	c. Increase or decrease gaps or continuities between spawning areas.	Change in occupancy of MaSAs that affects connectivity within the population.	A.1.c			
Goal B: 1. Maintain natural patterns of phenotypic and genotypic expression.	***a. Major life history strategies.***	Distribution of major life history expression within a population	B.1.a	Lowest score (highest risk)	B1 Mech. Score or Mean of B.1, B.2,B.3, and B.4, whichever is lower (higher risk)	
	b. Phenotypic variation.	Reduction in variability of traits, shift in mean value of trait, loss of traits.	B.1.b			
	c. Genetic variation.	Analysis addressing within and between population genetic variation.	B.1.c			
Goal B: 2. Maintain natural patterns of gene flow.	***a. Spawner composition.***	Proportion of natural spawners that are out-of-ESU spawners.	If two metrics rated as moderate, then high risk; otherwise lowest score (highest risk)	If two metrics rated as moderate, then high risk; otherwise lowest score (highest risk)		
		Proportion of natural spawners that are out-of-MPG spawners.				
		Proportion of hatchery origin natural spawners derived from a within MPG brood stock program, or within population (not best practices) program.				
		Proportion of hatchery origin natural spawners derived from a local (within population) brood stock program using best practices.				
Goal B: 3. Maintain occupancy in a natural variety of available habitat types.	***a. Distribution of population across habitat types.***	Change in occupancy across ecoregion types	B.3.a	B.3.a		
Goal B: 4.Maintain integrity of natural systems.	***a. Selective change in natural processes or impacts.***	Ongoing anthropogenic activities inducing selective mortality or habitat change within or out of population boundary	B.4.a	B.4.a		

Generating a Final Population-level Risk Rating

The primary purpose of our population level criteria is to identify populations performing at viable or highly viable levels. Our MPG level criteria require that a minimum number of the historical populations within a particular MPG be rated as viable or highly viable. In addition, the MPG criteria require that the other populations in a MPG be maintained at levels sufficient to provide for ecological functions and to preserve options for ESU recovery.

We integrate all four VSP parameters using a simple matrix approach as a framework (Figure 15). We base our ratings of the overall status of each population two composite metrics. The A/P metric combines the abundance and productivity VSP criteria (McElhany et al. 2001) using a viability curve. The second composite metric (SS/D) integrates across twelve measures of spatial structure and diversity. Determining if the remaining populations in an MPG are satisfying the maintained criteria requires additional considerations described below.

Viable and Highly viable populations are rated directly as specific combinations of A/P and SS/D risk ratings (illustrated in Figure 15). The composite A/P and SS/D metrics are expressed relative to a 5% risk of extinction within 100 years. Populations with a Very Low rating for A/P and at least a Low rating for SS/D are considered to be "Highly Viable." Populations rated at Moderate or High risk for A/P or High risk for SS/D have a risk of extinction greater than 5% and are not considered Viable. Although SS/D status is more difficult to quantify, populations rated at high risk against our composite SS/D criteria are not consistent with long-term persistence and viability.

		Spatial Structure/Diversity Risk			
		Very Low	**Low**	**Moderate**	**High**
Abundance/ Productivity Risk	**Very Low** (<1%)	**HV**	**HV**	**V**	M*
	Low (1-5%)	**V**	**V**	**V**	M*
	Moderate (6 – 25%)	M*	M*	M*	
	High (>25%)				

Figure 15. Matrix of possible Abundance/Productivity and Spatial Structure/Diversity scores for application at the population level. Percentages for abundance and productivity (A/P) scores represent the probability of extinction over a 100-year time period. Cells that contain a "V" are considered Viable combinations. "HV" indicates Highly Viable combinations. Shaded cells do not meet criteria for Viable status—darkest cells are at greatest risk. Cells designed as "M*" are candidates for maintained status.

The ICTRT criteria require a minimum number of populations within an MPG at or above viable status, with additional MPG populations maintained at sufficient levels to provide for ecological functions and to preserve options for ESU recovery. Maintained populations contribute to the ecological functioning of an ESU in several ways. The productivity of habitats and populations is dynamic and changes over time (Reeves et al. 1995). As a result, over a number of years source populations with higher productivities may exchange roles with sink populations with lower productivities in response to those changes (McElhany et al. 2000). The cumulative productivity across populations within an MPG should not fall below replacement (i.e. maintained populations should not serve as significant population sinks) (McElhany et al. 2000, Holmes and Semmens, 2004, Gunderson et al. 2001). In addition, if a catastrophe impacts one or more of the functioning viable populations within the MPG, the other populations will need to be at sufficient levels so that they can replenish those populations lost to or affected by the catastrophe. Maintained populations can also serve as genetic or demographic “stepping stones” between populations allowing natural patterns of gene flow and dispersal.

Maintained populations can also serve as buffers against uncertainty in the ICTRT population and MPG criteria. Ensuring that the less than viable populations meet maintained standards reduces the risk for the MPG. For example, an MPG with ½ the populations at viability and the remainder meeting maintained standards is at lower risk than an MPG with one or more populations at high risk. Additionally, having populations meet maintained standards should preserve recovery options in the event that efforts to recover other populations to viable levels fail.

Populations with specific combinations of A/P and SS/D ratings are candidates for Maintained status (Figure 15). However, it is difficult to capture all of the necessary attributes to meet the objectives for maintained populations in a simple set of integrated A/P and SS/D risk ratings. In general, populations with moderate abundance and productivity risk levels near 25% with high year-to-year variability or populations with high risk for multiple SS/D factors are less likely to be considered Maintained. A primary consideration in setting an abundance objective population in the smallest size category (Basic) would be uncertainty in current estimates of abundance and productivity. Given the levels of uncertainty in estimating recent geomean abundance and productivity, the abundance objectives for Basic populations should exceed 250 spawners to be designated as Maintained status. Populations classified in any of the three largest size categories should be at abundance levels not less than 500, and will likely require average abundance levels approaching minimum threshold values to address demographic and genetic considerations.

For each MPG, candidate populations should be reviewed individually and in context with the other populations against the above principles. Our use of a maintained population category is intended to result in similar contributions to persistence at the MPG level as would be achieved by meeting the Lower Columbia Willamette TRT requirements for a minimum average persistence score across populations within an MPG, and the Puget Sound TRT recommendation for “sustained” populations (PSTRT, 2002).

Monitoring and Evaluation

To provide general guidance for monitoring and evaluation, we identified improvements for current data collection and techniques to assess population status relative to the viability criteria. This section describes major data deficiencies but does not describe the specific sampling approaches needed to improve data quality. We highlighted major data deficiencies at the ESU/MPG level, however there are likely other population specific data needs that may be critical to viability assessment that we have not identified. In general, there were fairly large gaps in information for steelhead populations and the quality of information was generally poorer than for Chinook populations. We did not identify other M & E needs for limiting factors and action effectiveness, in this report. Key information gaps for conducting population level viability assessments include:

Abundance/Productivity:

1. Snake River steelhead population specific abundance and productivity data:
   A majority of populations had little or no recruit/spawner information to assess abundance and productivity criteria; most status assessments relied on a Snake River aggregate (Lower Granite) data set. Population level assessments for steelhead can be difficult given environmental conditions at the time of spawning, the potential distribution across stream drainages, etc. Alternative techniques should be considered (e.g., redd based surveys, weir counts combined with juvenile surveys, etc), incorporating probabilistic sampling protocols for estimating abundance.

2. Snake River steelhead population specific hatchery fraction and age structure data: A majority of populations had inadequate or no hatchery fraction information to assess abundance and productivity criteria. In addition, there is inadequate data to estimate the number of hatchery spawners in the aggregate recruit/spawner analysis. A majority of populations had no or inadequate age structure information to assess abundance and productivity criteria; most status assessments relied on a Snake River aggregate (Lower Granite) data set.

3. Upper and Mid Columbia Steelhead population abundance and productivity data: Most population abundance estimates are derived from standard index redd count surveys. Upper Columbia and Yakima population abundance are estimated from aggregate dam counts and population specific levels are apportioned using limited radio tag data. Abundance estimates need to be conducted using probabilistic sampling protocol for either redd counts or tagging studies.

4. Upper and Mid Columbia steelhead population specific hatchery fraction and age structure data: A majority of populations had inadequate hatchery fraction information. We used MPG aggregate hatchery fraction for most populations. Abundance and productivity assessments would improve with more detailed population level hatchery fraction data. A majority of populations had inadequate age structure information. Typically, average MPG aggregate age structure from a few years of data was used in most cases for the population level.

5. SARs and juvenile productivity estimates for all Chinook ESUs and steelhead DPSs: Improve or collect information on SARs and juvenile productivity (i.e. smolts per spawner). SARs are essential for taking into account variability in survival during smolt outmigration and marine life stages in evaluating A&P criteria. The goal is to estimate SARs that are representative at the population level. There are a number of approaches to accomplish estimating these SARs (e.g. marking wild or hatchery smolts or estimating natural origin smolts and adult production). In addition, measures representing survival from spawning to outmigrating smolts would aid in partitioning productivity between freshwater and marine life-stages.

6. Population level effects of hatchery spawners on natural productivity for all ESUs and DPSs: For populations with hatchery spawners, develop representative estimates of the effects of hatchery spawners on population level productivity. Topics of interest include the effect of hatchery spawner contributions to the average natural productivity of a population and the relative effectiveness of hatchery spawners. In combination with adequate estimates of the relative levels of hatchery fish contributing to natural spawning for a particular population, this information would allow for more representative estimates of current and potential natural productivity levels..

Spatial Structure and Diversity

1. Steelhead populations spawner distribution and habitat preference data: Many of populations had inadequate spawner distribution information to assess spatial structure and diversity criteria. In addition, estimates of historical distribution are dependent upon habitat preferences derived from available empirical studies. Those studies are limited in scope and number. Additional information on habitat/steelhead preference or production relationships could improve the assessment of steelhead populations against SS/D criteria.

2. Phenotypic characteristics for populations in all ESUs/DPSs: Little information was available to assess phenotypic changes. Representative estimates of current morphological, life history or behavioral traits are not available for many populations. Additional analysis of relationships between habitat characteristics and phenotypic traits would improve the ability to assess changes from historical patterns at the population level.

3. Steelhead genetics information, particularly for Upper Columbia and Mid Columbia populations: Genetic baseline information and periodic follow-up surveys specifically designed to evaluate the level of variation or differentiation among subcomponents within populations and among populations. Periodic follow-ups would support evaluation of responses to management actions designed to promote restoration of natural patterns of population structure.

4. Snake River Fall Chinook genetics sampling information allowing evaluation of population substructure: Establishing a baseline coupled with periodic future follow-up

efforts would generate information for evaluating the impacts of management strategies on population substructure.

5. Spawner composition for steelhead populations with hatchery spawners: Collect specific spawner composition information including proportion and source of hatchery spawners. Information on the relative distribution of hatchery spawners among production areas within populations would also improve the ability to assess status against ICTRT spatial structure criteria.

6. Selective mortality effects for populations in all ESUs/DPSs: Little information was available to assess selective mortality resulting from differential impacts of human induced mortality. Additional information is needed to better assess human induced mortality effects in each of the four Hs (habitat, hatcheries, harvest and hydropower)

There is considerable variability in the quality and quantity of information to conduct viability assessments for Interior Columbia River salmon and steelhead populations. We have identified fairly large gaps in information for steelhead populations and the quality of information was generally poorer than for Chinook populations. We believe improving the quality and quantity of data for the metrics and populations we identified above is essential for monitoring future change in population status relative to viability criteria.

Conclusions: Applying the ICTRT Viability Criteria

Our viability criteria reflect the hierarchical structure of Interior Columbia ESUs. ESU viability is a product of the viability of major population groups (MPG) and, in turn, the populations within them. Ecological and genetic patterns inherent in the distribution of populations within these levels contribute to the evolutionary history of the species. The viability of an ESU cannot be evaluated without first understanding the viability of these component building blocks. Thus our primary goal under this hierarchy has been to describe ESU viability through assessment of population extinction risks which consider abundance, productivity, spatial structure and diversity. Abundance plays an important role in our viability criteria, since abundance is a key element of extinction risk. However, it is important to recognize that a measure of average abundance alone is not sufficient for viability. The population and ESU level trends, distribution patterns and evolutionary potential (diversity) all contribute to ESU evolutionary and ecological functionality. Our criteria at all levels seek to tie viability to the primary drivers of evolutionary and ecological functionality.

Previous drafts of the ICTRT viability criteria were made available to provide guidance to regional recovery planning efforts that were ongoing concurrently with the development of these viability criteria. Early versions of the criteria were tested on some populations and refined based on lessons learned from the tests and input from regional recovery planners. The specific set of objectives and the particular measures associated with each component of our criteria have not changed. In some cases, the definition of certain risk levels in terms of a particular metric have been modified to facilitate more objective and consistent application of the criteria as well as to reflect new or better information as it became available. In addition, updates to the analyses used to estimate historical production capacity have resulted in changes in the assignment of some populations to a historical size category.

The biological viability criteria described in this report are developed to inform long-term regional recovery planning efforts and delisting criteria. Given that intent, we worked to express the criteria in objective, measurable metrics. This provides a level of transparency that facilitates critical review and future refinements. In addition, the criteria we used to express viability facilitate the development of effective recovery strategies by focusing attention on specific, often spatially explicit, biological conditions or processes. For example, our criteria include quantitative metrics expressed in terms of the current distribution of spawners relative to spatially explicit maps of historical production potential within a population. We provide examples of the relative risk associated with a range of general spawning area configurations. The descriptions of risk associated with alternative configurations provide recovery planners with an objective basis for targeting actions to address that component of viability. Our abundance and productivity criteria were designed to be used, in combination with current assessments, to inform recovery planning efforts as to the relative magnitude of changes in survival and habitat capacity needed to achieve viable status. They can also provide insight into whether productivity alone, or both productivity and capacity might need to be improved. We provide population specific estimates of the relative improvements in productivity and abundance required based on current assessments in a separate report (ICTRT, 2006).

We discuss some of the key uncertainties and their implications relative to viability criteria in this report and provide guidance for addressing uncertainty. This will allow both scientists and policy-makers to include this uncertainty as they consider these criteria. For example, we provide options for directly including sampling uncertainty into estimates of current abundance and productivity parameters. For some populations, additional data or analyses may provide results that can improve current status assessments. We included some guidance for considering additional analyses in assessing status in terms of particular viability metrics (e.g., estimating population level productivity). Where alternative data or analyses are used for comparison, a clear rationale should be provided.

The biological viability criteria described in this document lay out population, MPG and ESU-level characteristics that, given currently available information, would be associated with persistence of salmonid ESUs for the foreseeable future. Two groups of TRT products will be forthcoming that rely heavily on these criteria. First, we are currently conducting this type of current status assessment for all populations in the Interior Columbia, and intend to compile the assessments in a salmon and steelhead “atlas.” Drafts are available on our website: http://www.nwfsc.noaa.gov/trt/trt_current_status_assessments.cfm.

Second, we are conducting life-cycle modeling to assess the likely impact of different climate and hydropower scenarios on population status with respect to these criteria. Preliminary reports are also available on our website: http://www.nwfsc.noaa.gov/trt/trt_ic_viability_survival.cfm.

We have included two population viability assessments, Wenatchee River Spring Chinook Salmon and Umatilla River Summer Steelhead, as attachments to this document to serve as examples of applying our population level viability criteria. These examples illustrate how current risk ratings for individual metrics can be estimated using the guidance provided in this report. In addition, these examples illustrate how to integrate across the metric level assessments to generate an overall risk rating for a particular population. The population-level assessments provide the basis for evaluating viability at the next hierarchical level, the MPG. For MPGs with several populations, there typically are several scenarios or combinations of populations that would satisfy our MPG-level viability criteria. Those scenarios are described in Appendix G. For example, the John Day River MPG is one of four MPGs in the Mid-Columbia Steelhead ESU. This MPG consists of five populations. Applying the MPG-level viability criteria related to population size described in this report, the John Day River MPG could be rated at viable status if the Lower Mainstem John Day River, North Fork John Day River, and either Middle Fork John Day River or Upper Mainstem John Day River populations meet the criteria for a viable population. In addition, the remaining two populations in the MPG would need to be rated as maintained using the guidance provided in this report. Based on the draft population status reviews, the North Fork population is rated Highly Viable but none of the other populations in the MPG satisfy the criteria for a viable population. Therefore, this MPG does not currently meet viability criteria. The John Day and the other three MPGs would need to meet viability criteria for the Mid-Columbia Steelhead ESU to be rated as viable. The scenarios or combinations of populations that would be consistent with our MPG and ESU-level criteria for all ESUs are explicitly described in Attachment 2.

The ICTRT viability criteria describe biological characteristics for an ESU, MPG, and component populations consistent with a high probability of long-term persistence. The criteria were designed so that an ESU would be able to survive adverse fluctuations from average environmental conditions while maintaining long-term adaptive potential, given our current understanding of population and metapopulation processes. The TRT viability criteria metrics are expressed as specific values that can inform setting quantitative biological objectives for long-term recovery planning. The metrics, in combination with limiting factors assessments, can be used in targeting and sizing recovery planning strategies on factors that have a high potential for improving the status of the component populations of ESUs. The criteria can also be directly applied or readily adapted to assess the potential risk implications of proposed implementation strategies.

Literature Cited

USFWS 2002. Fall chinook salmon spawning ground surveys in the Snake River basin upriver of Lower Granite Dam, 2002. Annual Rept. to Bonneville Power Administration. USFWS, Nez Perce Tribe Fisheries Dept., & Idaho Power Company. Sept. 2003. 49 p

Beacham TD, Murray CB (1988) A genetic analysis of body size in pink salmon (*Oncorhynchus gorbuscha*). Genome 30:31-35.

Bentzen, P. Olsen, J.B. McLean, J.E., Seamons, T.R. and Quinn, T.P. 2001. Kinship analysis of Pacific salmon: insights into mating, homing, and timing of reproduction. J. Heredity 92:127-136.

Bevan, D., J. Harville, P. Bergman, T. Bjornn, J. Crutchfield, P. Klingeman and J. Litchfield. 1994. Snake River Recovery Team: final recommendations to National Marine Fisheries Service. May 1994.

Boyce, M.S. 1992. Population Viability Analysis. Annual Review of Ecology and Systematics. **23**: 481-506.

Bryden CA, Heath DD (2000) Heritability of fluctuating asymmetry for multiple traits in Chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 57:2186-2192.

Brooks, BW et al. 2002. Critiques of PVA ask the wrong questions: throwing the heuristic baby out with the numerical bath water. Cons. Biol. 16(1): 262-263.

Burgner, R.L. 1991. The life history of sockeye salmon (Oncorhynchus nerka). In: C. Groot and L. Margolis (eds.), Life history of Pacific salmon. p. 3 -117. Univ. of British Columbia Press. Vancouver, B.C.

Chatter, J.C., V.L. Butler, M.J. Scott, D.J. Anderson, and D.A. Neitzel. 1995. A paleoscience approach to estimating the effects of climatic warming on salmonid fisheries of the Columbia River Basin. In R.J. Beamish (ed.) Climate change and northern fish populations. Canadian Special Publications Fisheries and Aquatic Science **121**:4899-496.

Connor, WP, A.P. Garcia, A.H. Connor, E.O. Garton, P.A. Groves and J.A. Chandler. 2001. Estimating the carrying capacity of the Snake River for fall chinook salmon redds. Northwest Science. 75(4): 363-371.

Cramer, S., R. Beamesderfer, P. Monk, K. Witty, A. Kalin, B. Lister and C. Ackerman. (2002) Mid-Columbia steelhead trout population viability assessment. Progress Rept. S. P. Cramer & Assoc.

Crossin, G. T., S. G. Hinch, A. P. Farrell, D. A. Higgs, A. G. Lotto, J. D. Oakes, and M. C. Healey. 2004. Energetics and morphology of sockeye salmon: effects of upriver migratory distance and elevation. Journal of Fish Biology **65**:788-810.

Dauble, D.D., T.P Hanrahan, D.R. Geist and M.J Parsley. Impacts of the Columbia River hydroelectric system on main-stem habitats of fall chinook salmon. N.Amer. J. Fish. Mgmt. 23:641-659.

Deriso, R.B., D.R. Marmorek, and I.J. Parnell. 2001. Retrospective patterns of differential mortality and common year-effects experienced by spring and summer chinook salmon (Onchorhynchus tshawsytscha) of the Columbia River. Can. J. Fish. Aquat. Sci. 58: 2419-2430

Dickerson, B.R., M.F. Willson, P. Bentzen, and T.P. Quinn. 2005. Heritability of life history and morphological traits in a wild pink salmon population assessed by DNA parentage analysis. Transactions of the American Fisheries Society. **134**: 1323-1328.

Dunham, J.B., and G.L. Vinyard. 1997. Incorporating stream level variability into analyses of site level fish habitat relationships: Some cautionary examples. Transactions of the American Fisheries Society. **126**: 323-329

Epifanio, J., Haas, G., Pratt, K., Rieman, B., Spruell, P. Stockwell, C., Utter, F., Young, W. 2003. Integrating conservation genetic considerations into conservation planning: a case study of bull trout in the Lake Pend Oreille-lower Clark Fork River system. Fisheries 28(8):10-24.

Fagan, W.F. 2002. Connectivity, fragmentation, and extinction risk in dendritic metapopulations. Ecology. **83**: 3243-3249.

Falconer, D. S. 1960. Introduction to quantitative genetics. Ronald Press, New York.

Flagg, T. A., C. V. W. Mahnken, and R. N. Iwamoto. 2004. Conservation hatchery protocols for Pacific salmon. American Fisheries Society Symposium **44**:603-619.

Frissell, C. A., W. J. Liss, C. E. Warren, and M. D. Hurley. 1986. A Hierarchical Framework for Stream Habitat Classification - Viewing Streams in a Watershed Context. Environmental Management **10**:199-214.

Gall GAE, Neira R (2004) Genetic analysis of female reproduction traits of farmed coho salmon (*Oncorhynchus kisutch*). Aquaculture 234:143-154.

Gallant, A. L., T. R. Whittier, D. P. Larsen, J. M. Omernik, and R. M. Hughes. 1989. Regionalization as a tool for managing environmental resources. EPA/600/3-89/060, U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

Garant D, Dodson JJ, Bernatchez L (2003) Differential reproductive success and heritability of alternative reproductive tactics in wild Atlantic salmon (*Salmo salar* L.) Evolution 57:1133-1141.

Gharrett, .J. and Smoker, W.W. 1993. A perspective on the adaptive importance of genetic infrastructure in salmon populations to ocean ranching in Alaska. Fisheries Research 18:45-58.

Good, T.P, R.S. Waples and P. Adams (editors). 2005. Updated status of federally listed ESUs of west coast salmon and steelhead. U.S. Dept. of Commer. NOAA Tech. Memo. NMFS-NWFSC-66, 598 p.

Groves, P.A. & J.A. Chandler. 1999. Spawning habitat used by fall chinook salmon in the Snake River. No.

Amer. J. Fisheries Management. 19:912-922.

Gundersen, G., E. Johannesen, H.P. Andreassen, and R.A. Ims. 2001. Source-sink dynamics: how sinks affect demography of sources. Ecology Letters. **4**: 13-21.

Hallerman, E. M. (Editor). 2003. Population Genetics: Principles and Applications for Fisheries Scientists. American Fisheries Society, Bethesda, Maryland, 458 pp.

Hamon TR, Foote CJ, Hilborn R, Rogers DE. 2000. Selection on morphology of spawning wild sockeye salmon by a gill-net fishery. *Transactions of the American Fisheries Society* 129: 1300-15.

Hand ford P, Bell G, Reimchen T. 1977. A gillnet fishery considered as an experiment in artificial selection. *Journal of the Fisheries Research Board of Canada* 34: 954-61.

Hankin, D.G., J.W. Nicholas, and T.W. Downey. 1993. Evidence for Inheritance of Age of Maturity in Chinook Salmon (Oncorhynchus-Tshawytscha). Canadian Journal of Fisheries and Aquatic Sciences. **50**: 347-358.

Hard JJ. 2004. Evolution of chinook salmon life history under size-selective harvest. In *Evolution Illuminated: Salmon and Their Relatives*, ed. AP Hendry, SC Stearns, pp. 315-337. New York, NY: Oxford University Press.

Healey MC. 1986. Optimum size and age at maturity in Pacific salmon and effects of size-selective fisheries. In *Salmonid Age at Maturity, Canadian Special Publication of Fisheries and Aquatic Sciences*, ed. DJ Meerburg, pp. 39-52. Ottawa, Ontario: Canadian Department of Fisheries and Oceans.

Healy, M.C. 1991. Life history of chinook salmon. pp. 311-393 in: Pacific Salmon Life Histories. C. Groot and L. Margolis eds. Univ. British Columbia Press. 594 p.

Heath, D.D., L. Rankin, C.A. Bryden, J.W. Heath, and J.M. Shrimpton. 2002. Heritability and Y-chromosome influence in the jack male life history of chinook salmon (Oncorhynchus tshawytscha). Heredity **89**:311-317.

Hendry, A. P., J. E. Hensleigh, and R. R. Reisenbichler. 1998. Incubation temperature, developmental biology, and the divergence of sockeye salmon (Oncorhynchus nerka) within Lake Washington. Canadian Journal of Fisheries and Aquatic Sciences **55**:1387-1394.

Hendry, A. P., and T. P. Quinn. 1997. Variation in adult life history and morphology among Lake Washington sockeye salmon (Oncorhynchus nerka) populations in relation to habitat features and ancestral affinities. Canadian Journal of Fisheries and Aquatic Sciences **54**:75-84.

Hendry, A.P., Berg, O.K. and Quinn, T.P. 1999. Condition dependence and adaptation-by-time: breeding date, life history, and energy allocation within a population of salmon. Oikos 85:499-514.

Holmes, E. 2001. Estimating risks in declining populations with poor data. Proc. Nat. Acad. Sci. 98(9): 5072-5077.

Isaak, D.J., R.F. Thurow, B.E. Rieman, and J.B. Dunham. 2003. Temporal variation in synchrony among chinook salmon (Oncorhynchus tshawytscha) redd counts from a wilderness area in central Idaho. Can. J. Fish. Aquat. Sci. 60: 840-848.

ISG. 1996. Return to the river: restoration of salmonid fishes in the Columbia River ecosystem. Independent Scientific Advisory Board report to the Northwest Power Planning Council. Document 96-6. Portland, OR.

Kallimanis, A.S., W.E. Kunin, J.M. Halley, and S.P. Sgardelis. 2005 Metapopulation extinction risk under spatially autocorrelated disturbance. Cons. Biol. 19: 534-546.

Kinnison, M. I., M. J. Unwin, W. K. Hershberger, and T. P. Quinn. 1998. Egg size, fecundity, and development rate of two introduced New Zealand chinook salmon (Oncorhynchus tshawytscha) populations. Canadian Journal of Fisheries and Aquatic Sciences **55**:1946-1953.

Lichatowich, J.A. & L. E. Mobrand. 1995. Analysis of Chinook salmon in the Columbia River from an ecosystem perspective. Res. Rept. prepared for Bonn. Power Admin. Project #92-18. 102 p.

Lipscomb, S. W. 1998. Hydrologic classification and estimation of basin and hydrologic characteristics of subbasins in central Idaho. U.S. Geological Survey Professional Paper 1604:1-49.

Lush, J. L. 1937. Animal breeding plans. Iowa State College Press, Ames, IA.

Lynch, M., and B. Walsh. 1998. Genetics and analysis of quantitative Traits. Sinauer, Sunderland, MA.

Mackinson, S.,, M. Vasconcellos and N. Newlands. 1999 A new approach to the analysis of stock-recruitment relationships: "model free" estimation using fuzzy logic. Can. J. Fish. Aquatic Sci. 56: 689-699.

McClure, M. et al, 2003. Independent Populations of chinook, steelhead and sockeye for listed Interior Columbia Basin ESUs. Draft. Interior Columbia Basin Technical Recovery Team Report. July, 2003

McElhany, P., T. Backman, C. Busack, S. Heppell, S. Kolmes, A. Maule, J. Myers, D. Rawding, D. Shively, A. Steel, C. Steward & T. Whitesel. 2003. Draft Willamette/Lower Columbia Pacific Salmonid Viability Criteria. March, 2003

McElhany, P. Ruckelshaus, M.H., Ford, M.J., Wainwright, T.C. and Bjorkstedt, E.P. 2000. Viable salmonid populations and the recovery of evolutionarily significant units. Technical Memorandum NMFS-NWFSC-42. United States Department of Commerce, National Oceanic and Atmospheric Administration. Seattle, WA.

Michielsens, C.G.J., and M.K. McAllister. 2004 A baysian hieracrchical analysis of stock-recruit data: quantifying structural and parameter uncertainties. Can. J. Fish. Aquatic Sci. 61:1032-1047.

Merz, R.A. 2001. Stock and recruitment: generalizations about maximum reproductive rate, density dependence, and variability usign meta-analytic approaches. ICES J. of Mar. Sci. 58:937-951.

Milks, D., Varney, M, Schuck, M. and Sands, N. J. 2005. Lyons Ferry hatchery evaluation fall chinook salmon annual report: 2001 and 2002. Rept. to USFWS Lower Snake River Compensation Plan Office. April 2005. 100 p.

Mobrand, L. Barr, J., Blankenship, L.; Campton, D., Evelyn, T., Flagg, T., Mahnken, C., Seeb, L.W., Seidel, P. Smoker, W. 2005. Hatchery reform in Washington state: principles and emerging issues. Fisheries “in press”

Montgomery, D.R. and Bolton, S.M. 2003. Hydrogeomorphic variability and river restoration. In: Wissmar, R. and Bisson, P. (eds.) *Strategies for restoring river ecosystems: sources of variability and uncertainty in natural and managed systems*. American Fisheries Society, Bethesda, MD pp. 39-80.

Morris, W.F. and D. F. Doak. 2002. Quantitative Conservation Biology: Theory and practice of population viability analysis. Sinauer Assoc. Inc. U.S.A. 480 p.

Mousseau, T. A., K. Ritland, and D. D. Heath. 1998. A novel method for estimating heritability using molecular markers. Heredity **80**:218-224.

National Marine Fisheries Service. (1995). Proposed Recovery Plan for Snake River Salmon. U.S. Dept. of Commerce. March, 1995.

NRC (National Academy of Sciences). 1996. Upstream: salmon and society in the Pacific Northwest. Committee on protection and management of Pacific Northwest anadromous salmonids. Washington, DC.

NRC. 1995. Science and the Endangered Species Act. Committee on scientific issues in the endangered species act. Commission on Life Sciences. National Research Council. National Academy Press. 271 p.

Olson, D. E., B. Spateholts, M. Paiya, and D. E. Campton. 2004. Salmon hatcheries for the 21st century: a model at Warm Springs National Fish Hatchery. American Fisheries Society Symposium **44**:581-598.

Omernik, J. M. 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers **77**:118-125.

Omernik, J. M. 1995. Ecoregions: a spatial framework for environmental management. Pages 49-62 *in* W. Davis and T. P. Simon, editors. Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making. Lewis Publishing, Boca Raton, FL.

Orsborn, J. F. 1990. Quantitative modeling of the relationships among basin, channel and habitat characteristics for classification and impact assessment. 16D, Timber/Fish/Wildlife Program. CMER Project.

PSTRT, 2002. Planning ranges and preliminary guidelines for the delisting and recovery of the Puget Sound Chinook Salmon Evolutionarily Significant Unit. Puget Sound Technical Team Report.

Available at Puget Sound Technical Team website.

Ricker WE. 1981. Changes in the average size and average age of Pacific salmon. *Canadian Journal of Fisheries and Aquatic Sciences* 38: 1636-56.

Reisenbichler, R.R. Utter, F.M. and Krueger. C.C. 2003. Genetic concepts and uncertainties in restoring fish populations and species. In: Wissmar, R. and Bisson, P. (eds.) *Strategies for restoring river ecosystems: sources of variability and uncertainty in natural and managed systems*. American Fisheries Society, Bethesda, MD, pp. 149-183.

Schrott, G. R., K. A. With, and A. T. W. King. 2005. On the importance of landscape history for assessing extinction risk. Ecological Applications **15**:493-506.

Tear, T. H., P. Kareiva, P. L. Angermeier, P. Comer, B. Czech, R. Kautz, L. Landon, D. Mehlman, K. Murphy, M. Ruckelshaus, J. M. Scott, and G. Wilhere. 2005. How much is enough? The recurrent problem of setting measurable objectives in conservation. Bioscience **55**:835-849.

Reiman, B.E. and J.B. Dunham. 2000. Metapopulations and salmonids: a synthesis of life history patterns and empirical observations. Ecol. Freshwater Fish. (9): 51-64.

Ryman, N. 1991. Conservation genetic considerations in fishery management. J. Fish Biol. 39 (Suppl A):211-224.

Ryman, N. and Laikre, L. 1991. Effects of supportive breeding on the genetically effective population size. Conservation Biology 5:325-329.

Svasand, T. (eds) *Stock enhancement and sea ranching: developments, pitfalls and opportunities.* Blackwell. Pp. 260-306.

Utter, F.M. 2001. Patterns of subspecific anthropogenic introgression in two salmonid general. Rev. Fish Biol. And Fish. 10:263-279.

Utter, F.M. 2005. Population genetics, conservation and evolution in salmonids and other widely cultured fishes: some perspectives over six decades. Rev. Fish Biol. Fish. 14:125-144.

Wang, J. and Ryman, N. 2001. Genetic effects of multiple generations of supportive breeding. Cons. Boil. 15:1619-1631.

Waples, R.S. 2004. Salmonid insights into effective population size. In: Hendry, A. and Stearns, S. (eds) *Evolution Illuminated: salmon and their relatives*. Pp. 295-314.

Waples, R.S. and Drake, J. 2004. Risk/benefit considerations for marine stock enhancement: a Pacific salmon perspective. In: Leber, K.; Kitada, S. Blankenship, H. and

Waples, R. S., R. G. Gustafson, L. A. Weitkamp, J. M. Myers, O. W. Johnson, P. J. Busby, J. J. Hard, G. J. Bryant, F. W. Waknitz, K. Neely, D. Teel, W. S. Grant, G. A. Winans, S. Phelps, A. Marshall, and B. M. Baker. 2001. Characterizing diversity in salmon from the Pacific Northwest. Journal of Fish Biology **59**:1-41.

Weigensberg, I. and D. A. Roff. 1996. Natural heritabilities: Can they be reliably estimated in the laboratory? Evolution 50: 2149-2157.

Wichmann, MC, K. Johst, M. Schwager, B. Blasius & F. Jeltsch. 2005 Extinction risk, coloured noise and the scaling of variance. Theor. Pop. Biol. 68: 29-40.

Whiteley, A. R., P. Spruell, and F. W. Allendorf. 2004. Ecological and life history characteristics predict population genetic divergence of two salmonids in the same landscape. Molecular Ecology **13**:3675-3688.

Yamamoto, S., K. Morita, I. Koizumi, and K. Maekawa. 2004. Genetic differentiation of white-spotted charr (Salvelinus leucomaenis) populations after habitat fragmentation: Spatial-temporal changes in gene frequencies. Conservation Genetics **5**:529-538.

Zabel, Richard, Mark Scheuerell, Michelle McClure and John G. Williams. 2006. The interplay between climate variability and density dependence in the population viability of Chinook salmon. Conservation Biology 20(1):190-200

Appendix A: Population Viability Curves for Interior Columbia Chinook and Steelhead ESUs

Interior Columbia Basin Technical Recovery Team
March 14, 2007

Background

The Interior Columbia Technical Recovery Team (ICTRT) adapted a modeling approach for generating viability curves (McElhany et al. 2003) as a means of expressing the productivity and abundance component of population level viability criteria. A viability curve is defined by a set of paired combinations of productivity and abundance values corresponding to a particular extinction or quasi-extinction risk level. The ICTRT viability criterion for abundance and productivity requires a combination that addresses considerations for demographic persistence, the maintenance of genetic integrity and resilience to localized catastrophic risks.

We incorporate a minimum abundance threshold corresponding to the relative size category of the target population to address this range of objectives (Figure A-1). The standard time frame for assessing risk of extinction used in our analyses was 100 years. Each combination of productivity and abundance on a particular viability curve projects to the same modeled risk of extinction over a 100 year period.

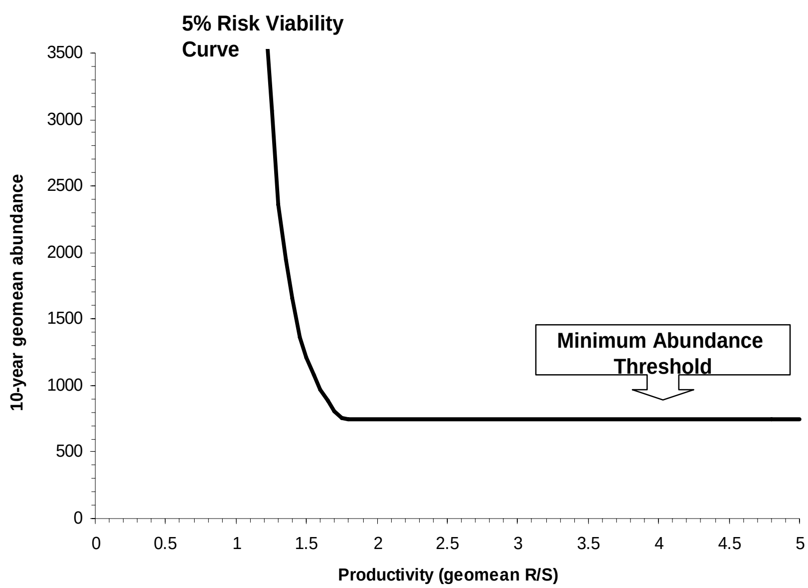


Figure A-1: Viability curve example. Curve represents combinations of abundance and productivity values associated with a 5% risk of extinction in 100 years, truncated to incorporate a minimum abundance threshold of 750.

The viability curve concept is adaptable, as the curves can be generated specific to a form of stock-recruit relationship and type of time series data available for a particular population or set of populations. In this example curve, abundance is expressed in terms of equilibrium spawning level and productivity as the expected geometric mean return per spawner at low to moderate abundance (the slope of the upward ascending limb of a Hockey-Stick function). In assessing the current status of a population against a viability curve, we recommend using a recent 10 year geomean of natural spawners as a measure of current abundance. Current intrinsic productivity should be estimated using spawner to spawner return pairs from low to moderate escapements over a recent 20 year period.

We developed two sets of ESU specific viability curves, each using a different measure of population growth rate. One set of curves expresses productivity in terms of return per spawner (to the spawning grounds). The alternative set of curves uses short term population growth rate (λ) as a measure of recent geomean productivity. The simple population growth rate based approach allows for assessments in circumstances in which the available data for assessing a population trend or abundance is limited and subject to high measurement error (Holmes, 2001). Fairly detailed annual spawner recruit data sets have been generated for most Interior Basin listed chinook populations and and many steelhead populations. Return per spawner based viability assessments can be directly adapted to accommodate large variation in annual abundance relative to potential capacity limitations as well as to autocorrelation in marine survival rates. We provide a detailed description of the derivation of the return per spawner based curves in the following sections, followed by a brief summary of adaptations of these basic steps to generate the population growth rate (λ) based viability curves.

In the following sections, we provide descriptions of the model we used to generate viability curves, descriptions of general and ESU specific input parameters, and a set of viability curves for each ESU. Representative estimates of year to year variability in return per spawner or population growth rates are key input parameters into the model used to generate population viability curves. We discuss key assumptions and uncertainties associated with curve generation and applications. We followed the basic approach for estimating variance and autocorrelation in production rates outlined in Morris & Doak (2002), adapting the approach to apply to time series of spawner to spawner return data sets.

We provide a brief summary of the use of viability curves in assessing current status. We used viability curves corresponding to a 25%, 5% and 1% risk of extinction in 100 years to define population level risks. Combinations of abundance and productivity falling below the 25% risk curve depicted in the chart (Fig. A-2) would be classified as at High risk. Combinations exceeding the 1% risk curve would be rated as at Very Low Risk. Abundance/productivity combinations falling between the 5% and 1% viability curves would be rated at Low Risk.

Under historical conditions, it is likely that most populations would have demonstrated combinations of intrinsic production potential and abundance well above the 5% Viability Curve. At the population level, recovery strategies should be targeted on achieving combinations of abundance and productivity above the threshold represented by the 5% viability curve. Estimates of current status will be based on sampling information and will therefore be influenced to some extent by sampling induced error and bias. We have provided some examples of approaches to directly incorporate provisions to minimize the potential for erroneously assigning a population to a relatively low risk status when the underlying risk may be high.

The last section of this attachment describes a sensitivity analysis of the effects on a curve of variations in each of the input parameters (variance and autocorrelation in productivity, age structure, and quasi-extinction threshold QET).

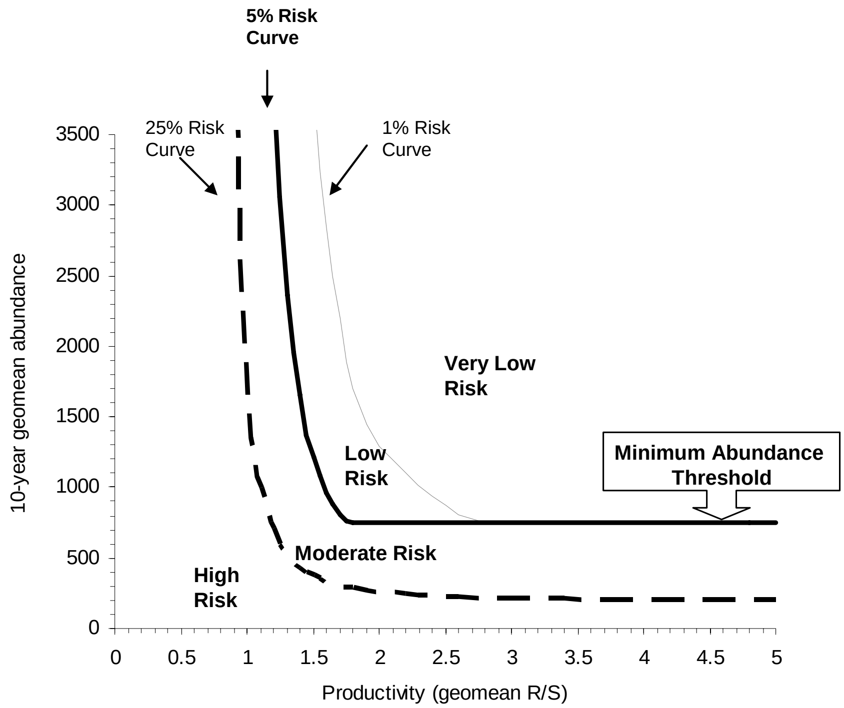


Figure A-2. Viability curve example. Curves represents combinations of abundance and productivity values associated with a 25%, 5% and 1% risk of extinction in 100 years, respectively. 5% and 1% curves truncated to incorporate a minimum abundance threshold of 750.

Viability Curve: Model Structure and Function

We used a stochastic cohort model to generate viability curves. The model generates a projected extinction risk given certain ESU-specific parameter estimates along with combinations of abundance and productivity. Additionally, the model includes an automated grid-search feature allowing the user to generate a viability curve corresponding to a selected risk level (e.g., 5% risk of extinction over a 100-year timeframe). We provide a detailed description of the mechanics of the model in this report.

The model operates on an annual time step. A model analysis consists of a minimum of 1000 iterations, each iteration being projected over at least 100 years. The cumulative results across the iterations are used to generate a probability of extinction corresponding to the input parameters for that analysis.

Stock-Recruit Function

The curves described in this report were generated using a hockey stick stock production function. We chose this function because it accommodates current status assessments based on simple measures of productivity at low abundance and production at capacity. It is also possible to express productivity and abundance/capacity in a viability curve in parameters in terms of the specific metrics in a particular stock-recruitment functions—e.g., Beverton Holt or Ricker curve a and b parameters. In most cases, data used to evaluate current status will be based on a relatively limited number of years. Uncertainty levels and bias in parameter estimates can be very large. Stock recruit function parameter estimates for relatively short data series that are based on fitting a standard function (e.g., Beverton Holt, Ricker or Hockey Stick) using a maximum likelihood or Bayesian fitting routine can contain substantial bias and/or uncertainty. These potential shortcomings are of less consequence if the available data series for a population is of sufficient length and/or if additional information is available to augment the trend data (e.g., environmental correlations, corresponding measures of juvenile production or smolt to adult survivals). Status assessments that use fitted stock recruit curve parameters as an index of current productivity should directly incorporate considerations for sampling induced errors and bias in their assessments.

Model Input Parameters

Two categories of input values are used in generating viability curves for application to Interior Columbia ESU populations. The first set included inputs that were common across all populations, regardless of ESU. Included in these generic inputs were the risk levels chosen for viability curves (e.g., 1%, 5%, and 25%) and the time period for assessing risk (100 years). This set also included values for extinction and reproductive failure thresholds as described below. The second set of parameters reflects characteristics of the specific populations within each ESU. Each population was assigned a minimum abundance threshold based on its estimated amount of historical spawning rearing habitat (see Attachment B). Population specific inputs included representative age at return proportions and a pair of parameters describing the expected variance and autocorrelation in annual return rates. The data sets used in generating population specific estimates of these parameters are included in population level current status assessments. Draft assessments are available at the ICTRT website. The ICTRT is developing an atlas of the current status assessments. That document will include a brief summary of regional methods for generating population specific estimates of annual abundance, age structure, etc.

Age at Return Distributions

We calculated average age distributions across available trend data sets for populations within each of the Interior Columbia listed salmonid ESUs. In some cases, population specific data sets were not available. If age composition estimates were available for aggregate returns including a population lacking a specific set of estimates, we assumed the aggregate estimate applied to that population.

Productivity: Variance and autocorrelation

One of our major objectives in this analysis was to identify variance and autocorrelation parameters representative of population productivity during rebuilding—a range that would include levels moderately above QET (50 spawners) to levels that would exceed the required equilibrium abundance thresholds specific to each population size category. We develop representative estimates of the variance and autocorrelation in annual return rate estimates for each of the listed Interior Columbia ESUs in this section. The estimates of annual variation in return rates were generated using population specific data sets and were averaged over a set of alternative stock-recruit functions (figure A-3).

Estimates for individual populations were based on relatively short data series subject to high levels of year to year variation. Therefore for those Interior Columbia ESUs represented by multiple populations (i.e., two stream type chinook and three steelhead ESUs), we averaged population level estimates of variance and autocorrelation across populations within ESUs to get representative sets of input parameters for generating viability curves. Population specific annual abundance data sets are described in Attachment B. We compiled brood year return estimates for the 20 most recent complete brood years for each data set.

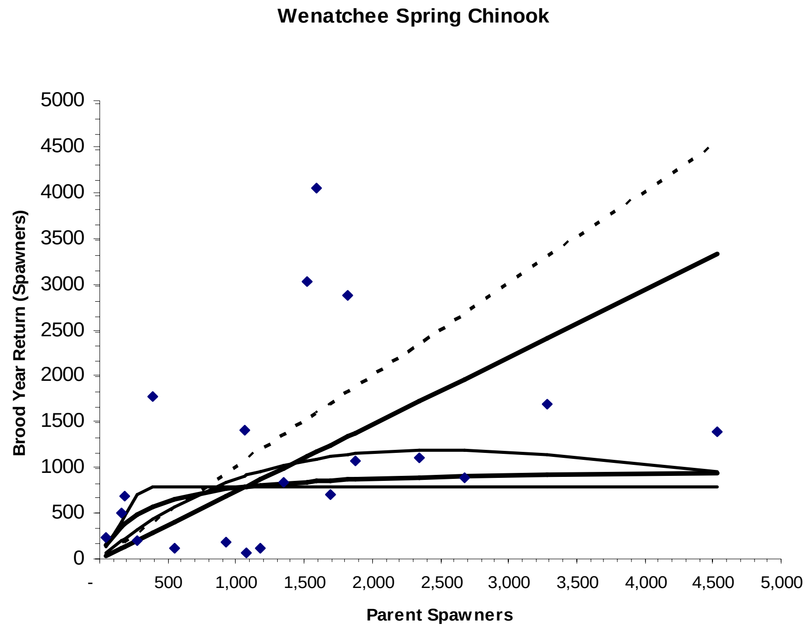


Figure A-3. Wenatchee River Spring chinook salmon population. Example of alternative stock-recruit functions (Random Walk, Hockey-Stick, Beverton/Holt and Ricker functions). Points are annual estimates of natural returns vs. total spawners in natural areas for brood years 1978 to 1999.

Differences in estimates between populations reflect the impacts of measurement error, departures from standard assumptions associated with fitting routines, etc. We considered a finer scale averaging (at the major population group level), but examination of the population level averages indicated more consistency at the ESU level.

We incorporated an autocorrelation parameter into the model used to generate viability curves based on results from our initial evaluation of representative trend data sets for Interior Columbia Basin Chinook and steelhead populations. We evaluated the time series of residuals from fitting a range potential stock recruit functions to the population specific data sets (Figure A-4). The annual residuals consistently demonstrated positive autocorrelation – that is, if the survival rate in a particular year was higher than average, there was a strong tendency for the survival in the following year to also be above average. Years that had relatively low survival rates tended to be followed by years with relatively low survival. The presence of autocorrelation in population growth rates can substantially influence projected extinction risks in population viability assessment models (Morris & Doak, 2002, Wichmann et al. 2005).

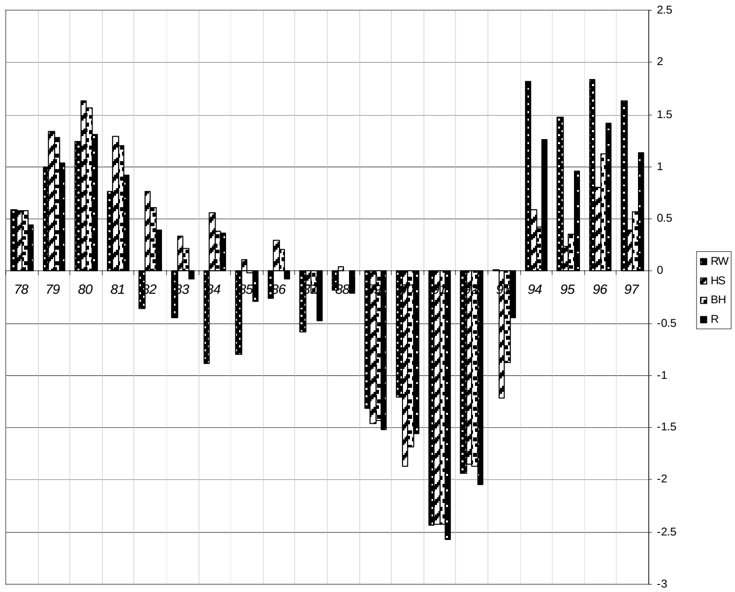


Figure A-4. Wenatchee River Spring Chinook salmon population. Deviations in annual return rates from predicted values using alternative stock/recruit functions.

We estimated simple one year lag correlation coefficients for the sequential series of residuals from fitting the basic stock-recruit functions to the individual trend data sets (Figure A-5). We limited our analysis to lag 1 correlations for several reasons: initial tests indicated lag 1 correlations were substantial and statistically significant; the data series we were evaluating were relatively short compared to the length required to estimate multiple year lag effects; and, incorporating lag 1 autocorrelation can effectively represent longer term cycles/patterns (e.g., Morris & Doaks, 2002).

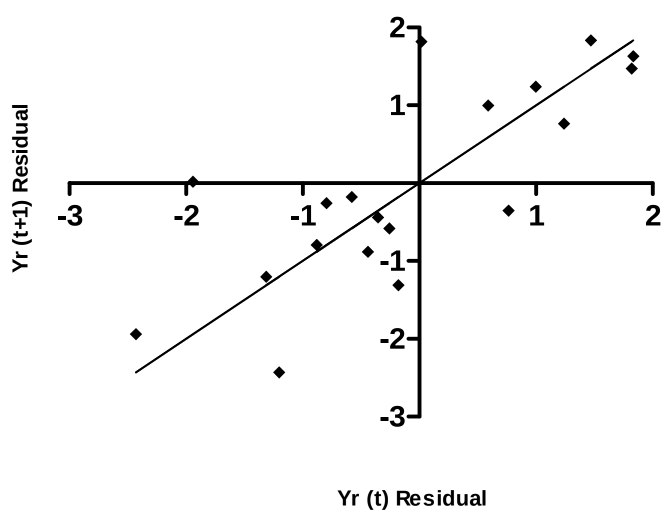


Figure A-5. Autocorrelation in annual variation in return rates. Wenatchee River Spring Chinook salmon population. Deviations in annual return rates from predicted values (Random Walk model). Points: year (t+1) vs. in year (t) residual deviations from predicted. Line represents 1:1 correspondence.

Quasi-Extinction Threshold

We evaluated model projections against a quasi-extinction threshold (QET) of 50 adult spawners per year over four consecutive years (generally corresponding to a brood cycle). A quasi-extinction threshold is defined as " ..the minimum number of individuals (often females) below which the population is likely to be critically and immediately imperiled." (Morris & Doaks, 2002; Ginsburg et al. 1982). We selected 50 as a QET based on four considerations; consistency with theoretical analyses of increasing demographic risks at low abundance, uncertainty regarding low abundance productivity of Interior Columbia ESU populations due to the paucity of escapements less than 50 spawners in the historical record, sensitivity analyses indicating that the probability of multiple very low escapements increases substantially as the QET approaches 1 spawner per year, and consistency with applications by the Puget Sound and the Lower Columbia/Willamette TRTs (McElhany et al. 2003, 2006; Puget Sound TRT, 200). We further discuss each of the rationale in the Population Abundance and Productivity section of our report on viability criteria (ICTRT, 2007).

Reproductive Failure Threshold

The QET is specifically expressed in terms of abundance over a four-year brood cycle. We also applied a Reproductive Failure Threshold (RFT) at the annual escapement time step in our model. In a given spawning year, production from an extremely low number of spawners are subject to decreases in reproductive success due to factors such as inability to find mates, random demographic effects, etc. In our viability modeling, we set production from a particular spawning year to zero if the adult escapement for that year was below the RFT. Initially, we set the RFT at the same value (on a per year basis) used in establishing a Quasi-extinction threshold (QET)—50 spawners. However, we have revised our estimate of the RFT appropriate for application to yearling type chinook and steelhead population model runs to 10 spawners after reviewing updated run reconstruction data sets for Interior Basins Spring/Summer Chinook populations and considering the potential for increases in sampling bias and heightened demographic risks as a function of extremely low abundance levels. We developed two simple analyses to inform setting the RFT at a number appropriate for Interior Basin chinook and steelhead populations. One analysis focused on the relative impact of sampling bias at low escapement levels, the other on a simplified model of demographic risk as a function of low escapements and multiple spawning sites.

Low Abundance Sampling Bias

Sampling related errors can substantially increase bias and variability in estimates of productivity derived for low spawning escapement levels. Our estimates of current intrinsic productivities for Interior Columbia Basin populations are based on annual population abundance data series. Natural returns are broken down into age components by applying a sampling based year specific age composition or an average age composition representative of the population. Year specific productivity estimates are then calculated by summing the returns by age corresponding to a particular brood year and dividing by the total parent escapement. Productivity estimates for extremely low spawning escapements in the data series can be biased upwards by sampling induced errors.

Annual spawner estimates for Interior Columbia Basin yearling type chinook populations are based on redd counts. At very low spawning levels, a single redd represents a substantial proportion of the total return. Annual return per spawner estimates are generated by total estimated returns at age for a given brood year by the parent spawning escapement in that brood year. Missing one or more additional redds at estimated total return levels of 2 to 10 spawners can result in substantial overestimates of spawner return rates.

Year to year variations in estimated spawning abundance is high. We developed a simple example of the potential impact on estimated productivity of year to year variability in abundance and the use of an average age composition to estimate brood year returns. The objective of the exercise was to evaluate the potential for bias in estimating productivity levels associated with extremely low spawning escapements (less than 100 spawners). We incorporated data from Interior Columbia Basin population abundance series into the assessment.

We averaged the relative ratios of low escapement year returns to returns in adjacent years across time series for Interior Columbia Basin population data sets. As an example, the estimated number of spawners in the Bear Valley population of spring/summer Chinook was 16 in 1995. The numbers of spawners estimated for 1994 and 1996 were 56 and 32, respectively. The ratios of the number of spawners in 1994 and 1996 to the estimate for 1995 were 3.5:1 and 2:1, averaging 2.8:1. We ordered spawning escapements and their relative ratios against adjoining return years and calculated median ratios across increments of 10 spawners (Figure A-6).

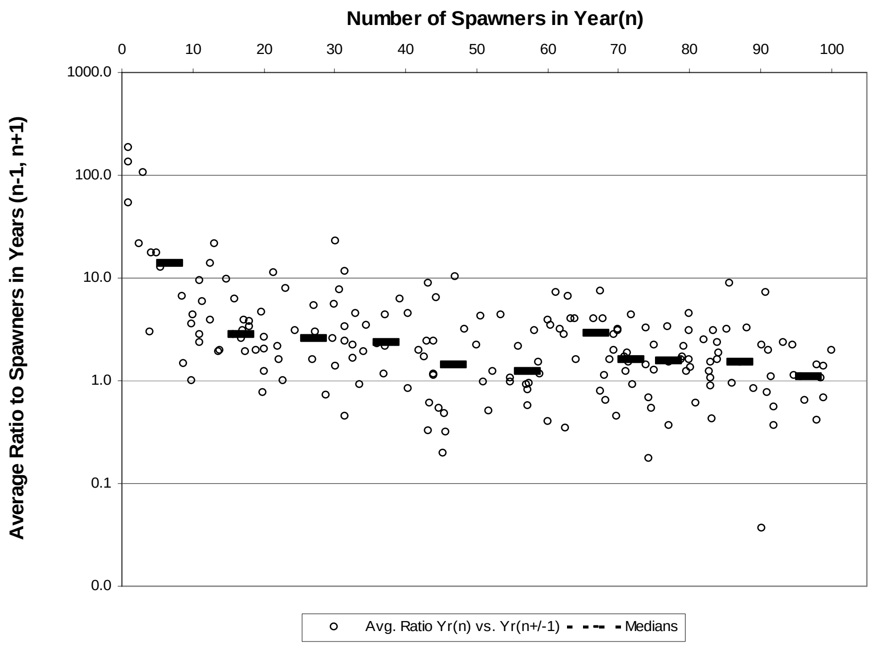


Figure A-6. Average ratios of spawner numbers in year$_n$ to spawner numbers in years$_{n\pm1}$ from Interior Columbia Basin population specific data sets. Ordered by the number of spawners in year$_n$.

Most of the low return levels in the data series were from relatively small populations in the Snake River Spring/summer ESU. For those series, the age information used to allocate natural returns to brood years with low parent escapement levels was an average for the population. For this exercise, we assumed an average age composition of 0.50 age 4 and 0.50 age 5 fish. A simple example will illustrate the level of bias in estimating productivity at low escapements that can arise from the combination of high variability in annual return rates and using average age composition data. Assume that a population data series includes a sequence of 100, 8 and 100 spawners in years 0, 1, and 2 and that the productivity for each of these years is 1.0. Equal proportions of the production from each brood year return at ages 4 and 5. In this scenario, 54 spawners would return in years 4 and 5. Applying the average age structure to year 4, an equal number of spawners (27) would be allocated to brood year 0 and to brood year 1. In this example, the same number of spawners (27) would be estimated as 5 year old spawners in year 6 and allocated to brood year 1. The total estimated returns from brood year 1 would be 55. The productivity from the escapement of 8 spawners in brood year 1 would be calculated as 55 divided by 8, or 6.9 returns per spawner—a substantial overestimate. In this example, estimates of annual productivities for escapements adjacent to the low escapement years would be systematically underestimated as a result of the misallocation of returns.

We evaluated the potential bias as a function of spawner level for escapements falling below 100 across spawning estimates from Interior Columbia population abundance data sets. We calculated median values across estimates grouped in increments of 10 and 25. We estimated the potential bias associated with the median ratios for each group under two different productivity assumptions: a) productivity in the adjacent brood years was equal; and b) productivity in the low escapement year was one 50% of the average productivity for the adjacent years in the series. The results of this simplified exercise indicate that the bias induced in estimates of productivity at low abundance can substantially inflate productivity estimates (Table A-1). The estimated impacts dropped rapidly as the number of spawners increased from 10 towards 50.

Misallocation of spawners to a particular brood year also affects productivity estimates at higher escapement levels. Median ratios of relative escapements in adjacent brood years approach one at higher escapement levels, indicating that the impact of misallocation by age would not result in a directional bias, but would largely translate into increased variance in estimated productivities.

Table A-1. Impact of bias in allocating returns on estimates of brood year specific productivities. Impact illustrated for two relative productivity scenarios: 1) actual productivity for low spawner escapement year equal to productivity for adjacent spawning years; and 2) actual productivity of low spawner brood year 50% of value for adjacent spawning years.

Number of Parent Spawners in $Year_n$	Median Ratio: Spawners(yr_n) to Spawners (yr_{n+1}, yr_{n-1}).)	Relative Bias: Estimated Productivity ($Year_n$)	
		$Year_n$ Productivity EQUAL TO $Year_{n-1,+1}$ Productivity	$Year_n$ Productivity 50% OF $Year_{n-1,+1}$ Productivity
2 to 10	15.8 : 1	8.40 X	16.3 X
11 to 20	3.1 : 1	2.05 X	3.6 X
21 to 30	2.7 : 1	1.85 X	3.2 X
31 to 40	2.3 : 1	1.65 X	2.80 X
41 to 50	1.5 : 1	1.25 X	1.75 X
50 to 75	1.7 : 1	1.35 X	2.20 X
76 to 100	1.5 : 1	1.25 X	2.00 X

Demographic Risk at Very Low Spawner Abundance

Given the production observed at low escapements, we also developed a simple stochastic simulation of demographics at very low population sizes to inform a revision of the RFT estimate. Spawning ground survey results indicate that spawning redds are often dispersed across several spawning sites within a population even at very low spawning densities. Under those circumstances the probability that one or more females may return to a site without male spawners. We set up a hypothetical population model assuming three spawning areas. We assumed that the average ratio of males to females was 1:1, with annual returns following a binomial distribution and that returning males and females would randomly distribute among the three spawning areas. We generated 1,000 iterations of the model for total spawning returns ranging from 2 to 16. We calculated the effective number of female spawners for each model iteration, defining an effective female spawner as a female return to a spawning area occupied by at least one male spawner. We averaged the proportion of effective female spawners across 1,000 iterations at each spawning level tested (Figure A-7). The expected proportion of effective female spawners decreased from greater than 0.90 to less than 0.80 as spawner numbers declined to below 10. Below this range, the proportion of effective spawners in this simple model decreased substantially as a function of decreasing return levels.

The results of these simple simulations supported setting an RFT of 10 spawners in the model for generating viability curves for yearling chinook populations. Upper Columbia steelhead populations also utilize tributary habitats for spawning and extended rearing. We applied the same RFT in developing viability curves for these populations. The primary spawning and rearing habitat for Snake River fall chinook is in the mainstem of the Snake River and the lower reaches of major tributaries. Spawning areas within the remaining population of Snake River fall chinook are distributed in relatively small patches across over 100 km of the mainstem Snake River. As a result, we retained a higher RFT of 50 spawners in generating a set of viability curves for application to the Snake River fall chinook population.

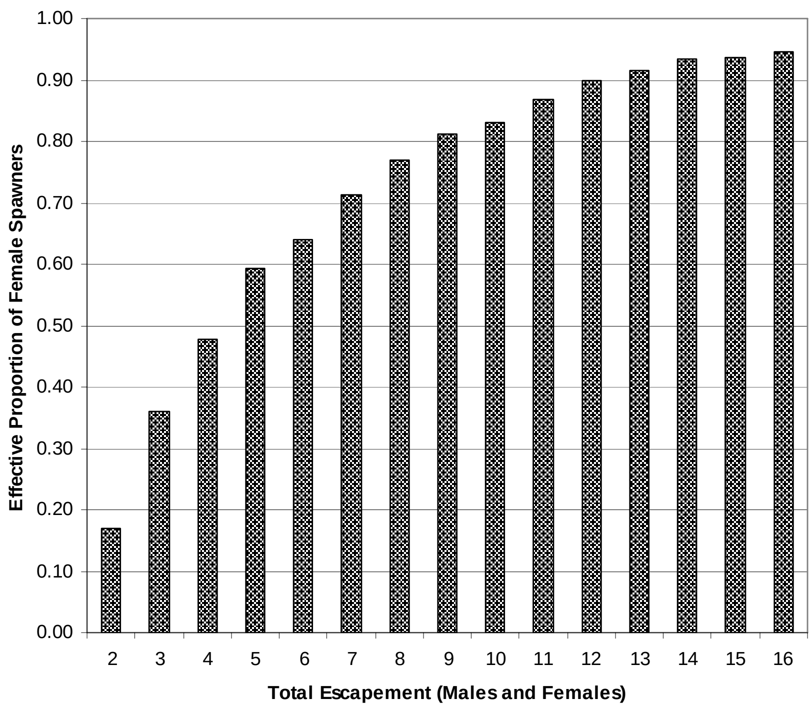


Figure A-7: Hypothetical three spawning area model. Proportion of returning females returning to a sub area with at least one male spawner present. Assumptions: 1:1 male to female ratio (binomial distribution), equal probabilities of migrating to any of the three areas. Effective proportion female spawners = effective female spawners/total female returns.

Model Mechanics

We used a cohort-based extinction risk model (described below) to calculate a standard set of viability curves for application to each ESU. The initial step in deriving a viability curve was the selection of a target risk level/time period, we generated curves corresponding to 1%, 5% and 25% risks of quasi-extinction over a 100 year timeframe.

Automated Grid Search Routine

Viability curves were generated by iteratively running the cohort model through a range of productivity and abundance combinations using an automated grid search routine. We used ESU-specific geomean return rate variance and autocorrelation estimates along with averaged age at return proportions as inputs into the model runs. We used the extinction risk model in conjunction with a binary search algorithm to estimate the equilibrium abundance associated with each individual productivity value in the series that yielded the target risk. The model can either be run in batch mode to search for the specific abundance levels associated with each productivity in an input series of values or to find the abundance corresponding to a particular productivity value.

For a given productivity, the model was run with the user-specified upper and lower abundance bounds, and extinction risk was evaluated for both runs. If the target extinction fell between the risks associated with both bounds, the algorithm would seed the model with the abundance halfway between the two previous values. The algorithm continued seeding the model using this “halfway” method until the resulting risk was within 7% of the target risk. At this point, 4000 iterations per run were used to minimize the risk of missing the appropriate abundance. Using 4000 iterations instead of the customary 1000 enabled a more stable and fine-scale risk analysis. Once an extinction risk within 0.5% of the target risk was found, the corresponding abundance value was recorded and the model moved on to the next productivity value in the series. After completing the entire series, the results were used to plot a rough viability curve. The derived values were used to seed the model for a final series of fine-scale iterations to improve accuracy and to smooth the curve.

Cohort Model Structure

User defined values were used to set average productivity and capacity terms specific to the stock recruit function used in the analysis. We used a form of the ‘Hockey Stick’ function in generating the ESU-specific population viability curves presented in this report. A simple modification to the model allows for running the analyses with a Beverton- Holt or a Ricker function (note that the productivity and capacity input values would need to be expressed in the corresponding metrics). The productivity and abundance parameters in the extinction risk model were expressed in terms that can be directly related to estimates that can be derived from abundance data series available for many Interior Columbia populations (equation A-1).

$$R(t) = A * MIN(S(t), SB) * \mathcal{E}(t) \qquad \text{eq. A-1}$$

Where:

R(t) = Expected number of adult returns to the spawning area in future years resulting from brood year escapement S(t).

S(t) = Parent year adult escapement.

SB = Spawner Breakpoint: number of spawners corresponding to breakpoint of hockey stick function.

A = Productivity: Estimated as geomean return/spawner at spawning abundance below SB.

ε(i) = *process error: random variable, lognormal distribution with a mean of 0, standard deviation of σ.*

Running the Model

Each modeled population projection is seeded with a series of five consecutive escapement values (years -4 to 0). For viability curve generation, the model was seeded with the spawner number being evaluated for the particular iteration of the grid search routine. The cohort model can also be used to generate an estimate of risk using population specific current abundance and productivity estimates. For a risk assessment of an individual stock, we used the five most recent spawning escapements as initial values.

Step 1—generating a population projection

The model steps through the escapement series, sequentially generating an estimate of production for each parent escapement. If the parent escapement value is below the user-defined reproductive failure threshold (RFT), the production from that brood year is set to zero. If the adult escapement exceeds the RFT, the model generates an initial production estimate using the embedded stock-recruit function with productivity and capacity terms based on the input values for the particular model run. The model applies an annual deviation to projected returns from each parent year based on a random draw from a normal distribution defined by estimates of ESU specific averages of variance and autocorrelation. The resulting production from spawning in year (t) is allocated to future returns by applying the user-defined average age distribution. Although age structure was kept static while generating the viability curves, the model was designed so that the user can add stochasticity to the annual brood year age distribution if desired.

The model incorporates autocorrelation into the annual stochastic error term adapting the approach described in Morris & Doak (2002). We used average variance and autocorrelation estimates corresponding to each ESU (see the Population Statistics section below). The model works in annual time steps. A run is initiated by calculating the expected production from the spawning escapement in year 1 and multiplying the result by a factor drawn from a lognormal distribution with mean of 0 and a standard deviation of σ, where σ is the average ESU value. The stochastic error term for year 2 and all subsequent production years is modified to incorporate autocorrelation:

$$\varepsilon(t) = \rho * \varepsilon(t-1) + \sigma' \qquad \text{eq. A-2}$$

where ρ is the simple correlation coefficient between sequential annual deviations from expected productivity calculated from the data series for the corresponding ESU and the term $\mathcal{E}$ (0,σ') represents the portion of the variance in the data series not accounted for by autocorrelation. The adjusted standard deviation in that term, σ', is calculated as:

$$\sigma' \cong \sqrt{\sigma^2} * \sqrt{1-\rho^2} \qquad \text{eq. A-3}$$

Model year 1 is the first year in each projection that is totally generated by the model (not an initial seed escapement). The model generates an estimate of adult escapement in year 1 by adding together the projected number of 5 year olds produced from the initial seed escapement in year (-4) and the projected number of 4 year old adults produced from initial seed escapement year (-3). The model repeats steps 1 and 2, generating a time series of at least 100 years.

Step 2—projection iteration

At the end of a 100+ year population projection, the model stores the series of annual abundance estimates in a temporary results file or virtual array. Under the basic set-up, 1000 projections (replicates) of 100+ years for each set of input parameters are generated during a model run. Each projection is based on the same input parameters (capacity and starting escapement values, variance, autocorrelation, and age structure), but reflects a unique combination of random draws from the distribution defined by the variance and autocorrelation input values. In other words, each projection for a particular set of model inputs represents an alternative potential future pattern in returns over a 100+ year time period that is consistent with that particular set of model inputs.

Step 3—Compiling a Risk Estimate

After 1,000 projections are accumulated, the model summarizes the results according to the specific risk target metrics input into the model. If the parent escapement from any four consecutive years leading up to (and including) the user-specified timeframe are all less than the QET, then the projection is counted as an extinction. We evaluated the projected risk of extinction over a 100-year period. Finally, the extinction risk for the entire run is calculated as the proportion of projections that were counted as extinct.

Minimum Abundance Thresholds

Populations of listed chinook and steelhead within Interior Columbia ESUs vary considerably in terms of the total area available to support spawning and rearing.

We add a minimum abundance threshold to our ESU specific viability curves corresponding estimates of the historical amount and complexity of tributary spawning habitat for a population. The minimum abundance thresholds were incorporated into the ESU specific viability curves to ensure that the full range of objectives defined for productivity and abundance are achieved, including the desire to maintain genetic characteristics and to maintain sufficient spawner densities in larger tributary habitats. A more detailed discussion of the rationale for the specific minimum abundance thresholds is included in the population viability criteria section of the ICTRT document and in Attachment B.

ESU-Specific Viability Curves

We generated sets of viability curves for application to populations within each of the Interior Columbia ESUs. We used ESU average estimates of variance and autocorrelation derived from representative trend data sets combined with minimum abundance thresholds specific to the general population size categories to generate curves. In addition to depicting the 5% risk of extinction threshold for evaluating population viability, the figures also include risk thresholds corresponding to a relatively high risk of extinction (10% and 25% in 100 years) and a lower risk level (1% in 100 years). We adapted the approach to accommodate the relatively limited amount of data available for Snake River Fall Chinook and Sockeye populations.

We analyzed the incremental and combined effects of filtering the data sets for factors that could inflate population level estimates of variability in return rates: multiple years with very low parent spawning levels, chronic high hatchery origin spawners, and incorporating a specific form of the spawner recruit relationship with relatively poor statistical fit across the data sets. The specific criteria used to screen populations for these factors are summarized in Table A-2.

Table A-2. Screening criteria used to develop representative estimates of variance and autocorrelation in productivity for input into ESU specific viability curve projections.

Factor	Criteria
1. Multiple spawnings at extreme low numbers	Most recent 20 year geomean of adult spawners less than 50 per year
2. Multiple years with high hatchery origin spawner proportions	Most recent 20 year average proportion hatchery (to spawning grounds) of greater than 30%.
3. High proportion and annual variability in hatchery proportion	High proportion screen plus standard deviation of hatchery proportion exceeds 30%
4. Worst fit statistical model (across populations)	Based on comparative AICc analyses within ESU populations. Drop model that most often scores lowest (by at least 2 AICc points) across populations within the ESU
5. Combination (1&2) multiple low and high potential hatchery influence	Apply criteria for factors 1 & 2
6. Combination (1&2) plus eliminate worst fit model (4)	Apply criteria for factors 1, 2 and 4

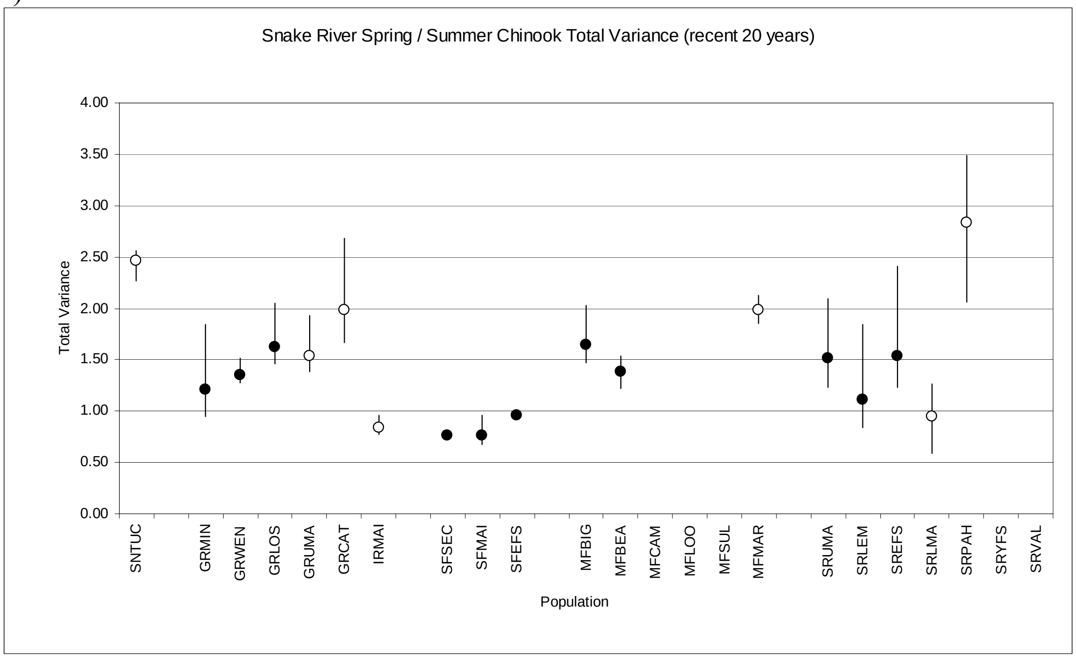
Snake River Spring/Summer Chinook ESU

We have developed 23 population specific data series for this ESU. Population level estimates of the variance and autocorrelation are depicted in Figure A-8. The average total variance and autocorrelation estimates based on all 23 population data series increased relative to the averages for the 12 data series available for the first draft of this analysis (ICTRT 2005a). Updates to the individual data series included in the original set accounted for a small component of the increase in both parameters (Table A-3). Most of the increase was due to the addition of the 11 new data series. The geomean in parent spawning levels were below 50 for five of the data series for this ESU, indicating multiple years with very low spawning numbers. The variance in return rates at very low spawning levels is likely significantly increased. Dropping those five data series from calculating the average resulted in reduced total variance and a moderate increase in average autocorrelation. Six of the twenty-three populations had relatively high inputs of hatchery origin fish into natural spawning across the 20 year time frame. Dropping those six populations from the analysis resulted in increased average total variance and autocorrelation. Excluding the s/r function with the worst fit across populations (Random Walk) resulted in reduced total variance and elevated average autocorrelations. Applying all three of the criteria drops ten population data sets from the analysis. The resulting average total variance is 1.24, approximately 10% higher than the estimate based on the original set of 12 population data series.

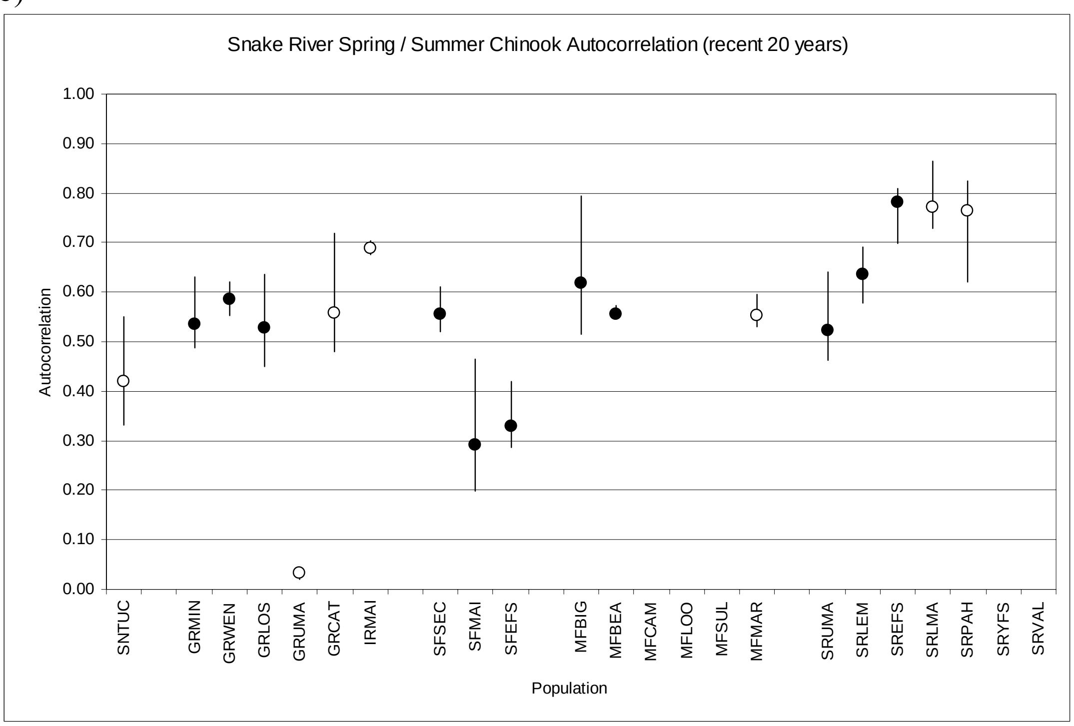
The viability curves generated for application to populations of Snake River spring/summer chinook within each of the four historical population size categories are depicted in Figure A-12a-d.

Figure A-8a-c. Population estimates of productivity (geomean brood year spawner to spawner return rates) statistics for the Snake River spring summer chinook ESU: a) total variance; b) autocorrelation; c) adjusted variance (after accounting for autocorrelation). Bars represent +/- 1 standard error. Filled symbols indicate population data series that met filters described in text.

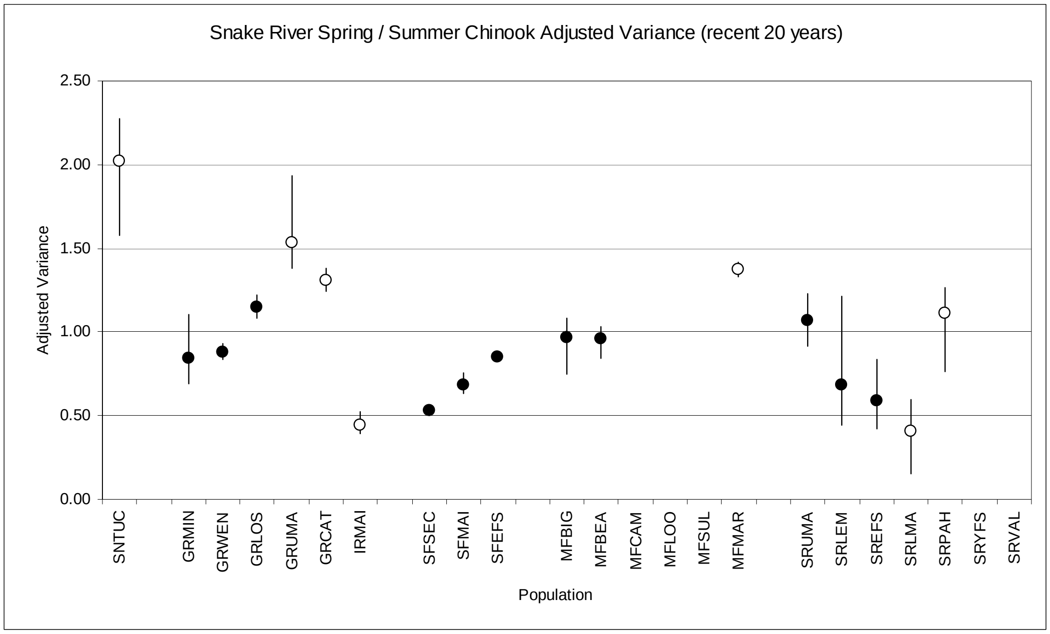
a)



b)



c)


Snake River Spring / Summer Chinook Adjusted Variance (recent 20 years)
Adjusted Variance
2.50
2.00
1.50
1.00
0.50
0.00
SNTUC
GRMIN
GRWEN
GRLOS
GRUMA
GRCAT
IRMAI
SFSEC
SFMAI
SFEFS
MFBIG
MFBEA
MFCAM
MFLOO
MFSUL
MFMAR
SRUMA
SRLEM
SREFS
SRLMA
SRPAH
SRYFS
SRVAL
Population

Upper Columbia Spring Chinook ESU

The original analysis included data sets for all three of the extant populations in the Upper Columbia spring chinook ESU. Updates to the data sets resulted in a small increase (roughly 3%) in total variance (Table A-3). Estimated average autocorrelation remained at the same value (0.68). None of the data sets were eliminated by the geomean population size and hatchery contribution tests. Eliminating the worst fit s/r model across the data series reduced the total variance to 0.95, approximately 3% below the original values.

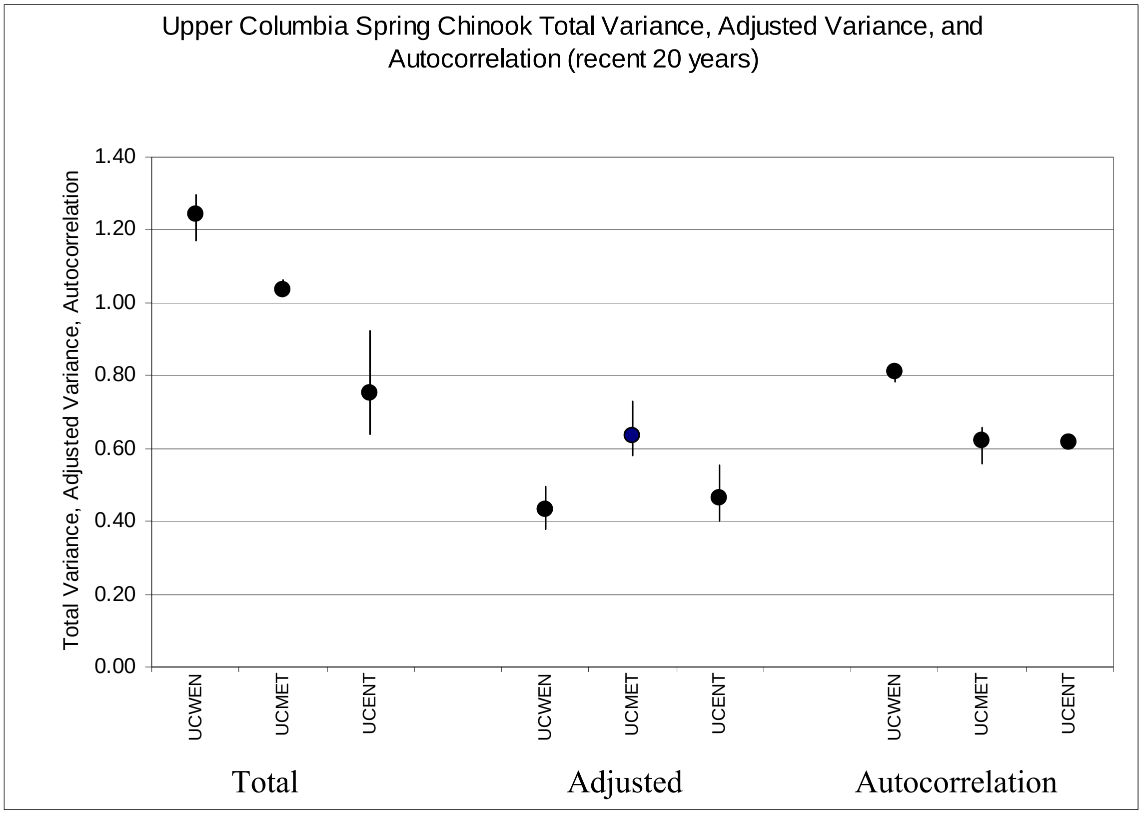


Figure A-9. Population estimates of productivity (geomean brood year spawner to spawner return rates) statistics for the Upper Columbia Spring Chinook ESU. Total variance, autocorrelation, and adjusted variance (after accounting for autocorrelation) are shown. Bars represent +/- 1 standard error. Filled symbols indicate population data series that met filters described in text.

Upper Columbia Steelhead ESU

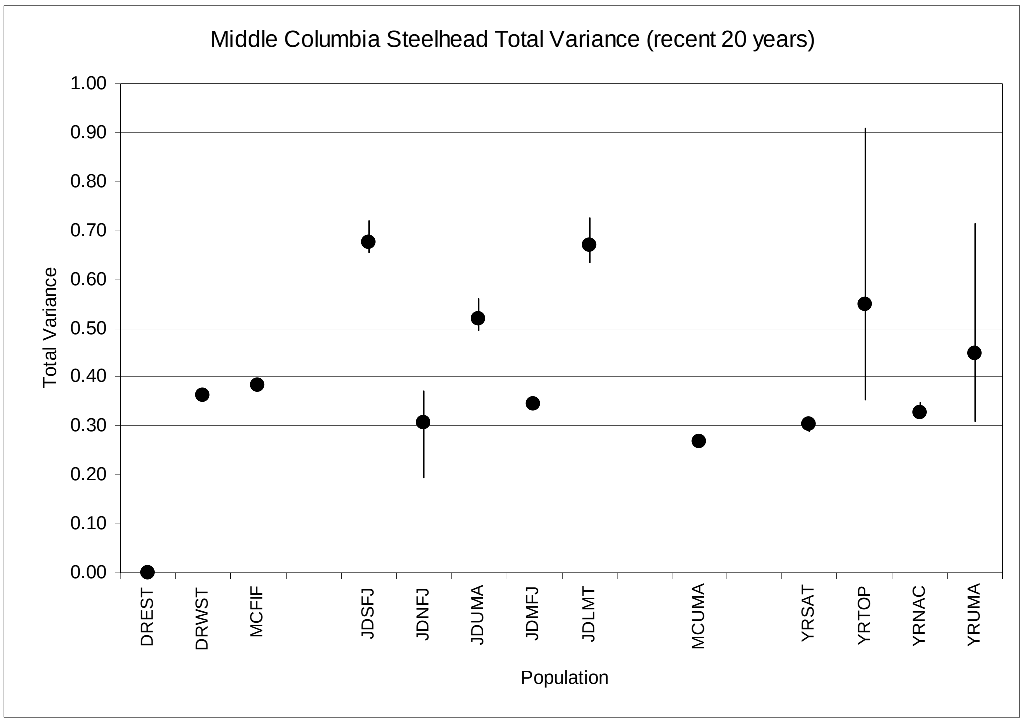
Since the ICTRT has little confidence in estimates of variance and autocorrelation for Upper Columbia Steelhead populations, combined estimates from the Mid-Columbia and Snake River steelhead ESUs were used in generating viability curves for the Upper Columbia ESU (Figures A-10 and A-11).

Mid-Columbia Steelhead ESU

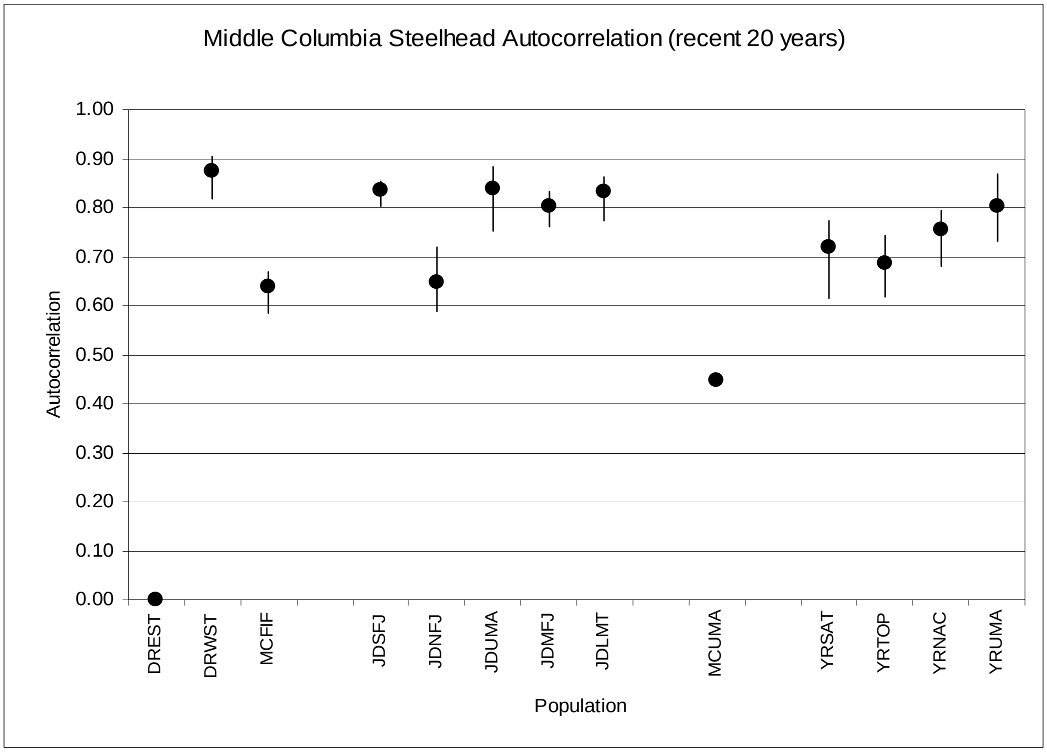
We generated variance and autocorrelation estimates using data sets representative of 13 Mid-Columbia steelhead populations (Figures A-10a-c). We calculated a set of average values across 12 of the data sets for use in generating a representative viability curve for application to populations within the ESU. We dropped the Deschutes River (Eastside) data set due to chronically high estimated proportions of hatchery origin fish on the spawning grounds.

Figure A-10a-c. Population estimates of productivity (geomean brood year spawner to spawner return rates) statistics for the Mid-Columbia Steelhead ESU. a) total variance; b) autocorrelation; c) adjusted variance (after accounting for autocorrelation). Bars represent +/- 1 standard error. Filled symbols indicate population data series that met filters described in text.

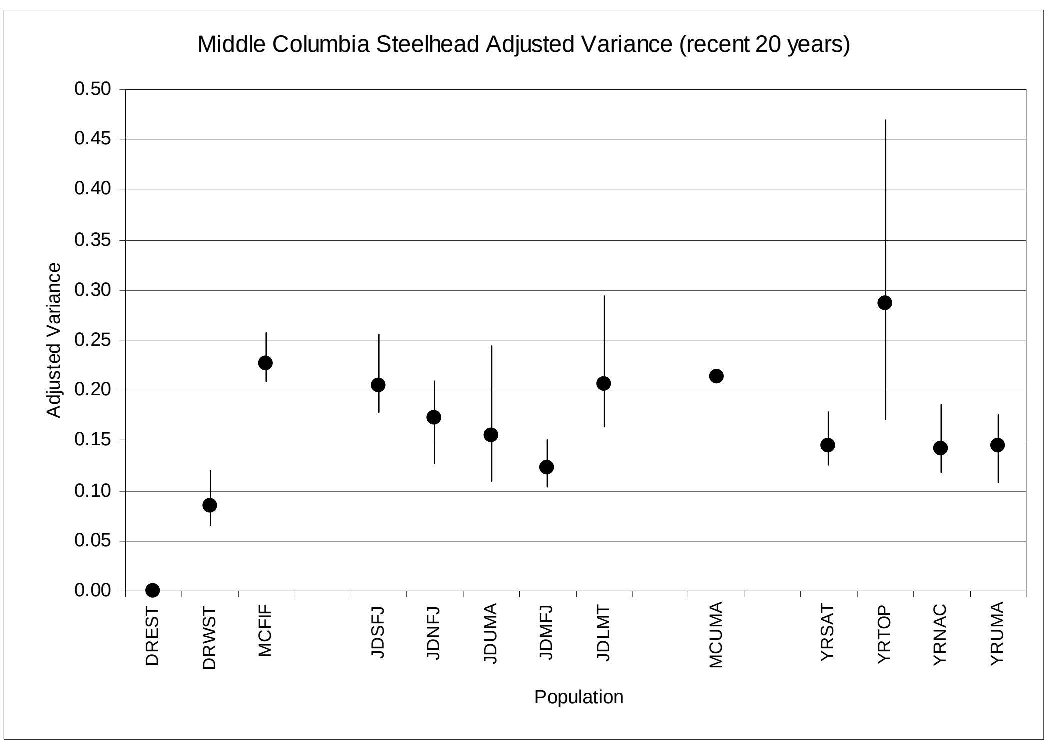
a)



b)


Middle Columbia Steelhead Autocorrelation (recent 20 years)
Autocorrelation
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
DREST
DRWST
MCFIF
JDSFJ
JDNFJ
JDUMA
JDMFJ
JDLMT
MCUMA
YRSAT
YRTOP
YRNAC
YRUMA
Population

c)


Middle Columbia Steelhead Adjusted Variance (recent 20 years)
Adjusted Variance
0.50
0.45
0.40
0.35
0.30
0.25
0.20
0.15
0.10
0.05
0.00
DREST
DRWST
MCFIF
JDSFJ
JDNFJ
JDUMA
JDMFJ
JDLMT
MCUMA
YRSAT
YRTOP
YRNAC
YRUMA
Population

Snake River Steelhead ESU

Population specific trend data sets area available for a relatively small proportion of populations in the Snake River Steelhead ESU. Three new population specific series have been developed in addition to the two original data sets used in previously reported ICTRT analyses. Four out of the five population specific trend series are in the Grande Ronde MPG and the adjacent Imnaha River. The only set specifically corresponding to returns to a particular location in the Idaho portion of the ESU was based on weir counts of fish returning to a section within the Little Salmon River population. Annual counts of wild and hatchery steelhead passing over Lower Granite Dam are available. These aggregate counts represent the combined returns to all populations and hatchery facilities above Lower Granite Dam and include the returns accounted for by the estimates described above. The Lower Granite counts can be broken down into A and B type steelhead runs (TAC ref). The populations with available trend series are all classified as Type A stocks. To complement the population specific trend data sets, we calculated return rate statistics (variance and autocorrelations) for average A and B run populations assuming that the returns not accounted for in the available population sets were distributed among the remaining populations proportional to intrinsic potential habitat.

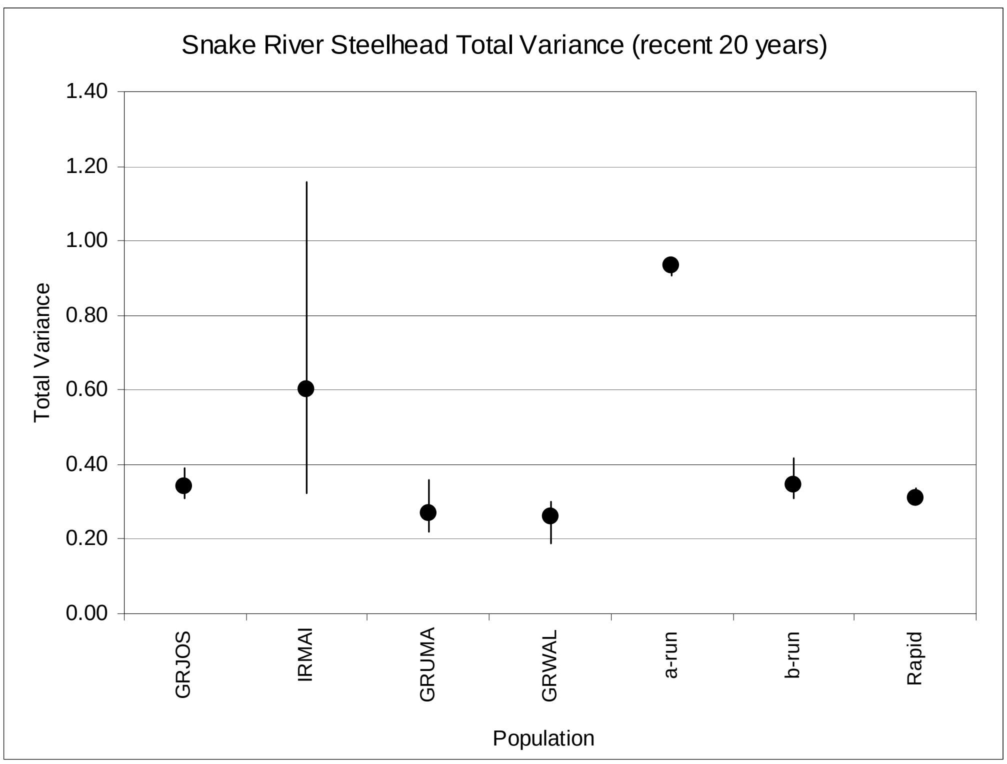
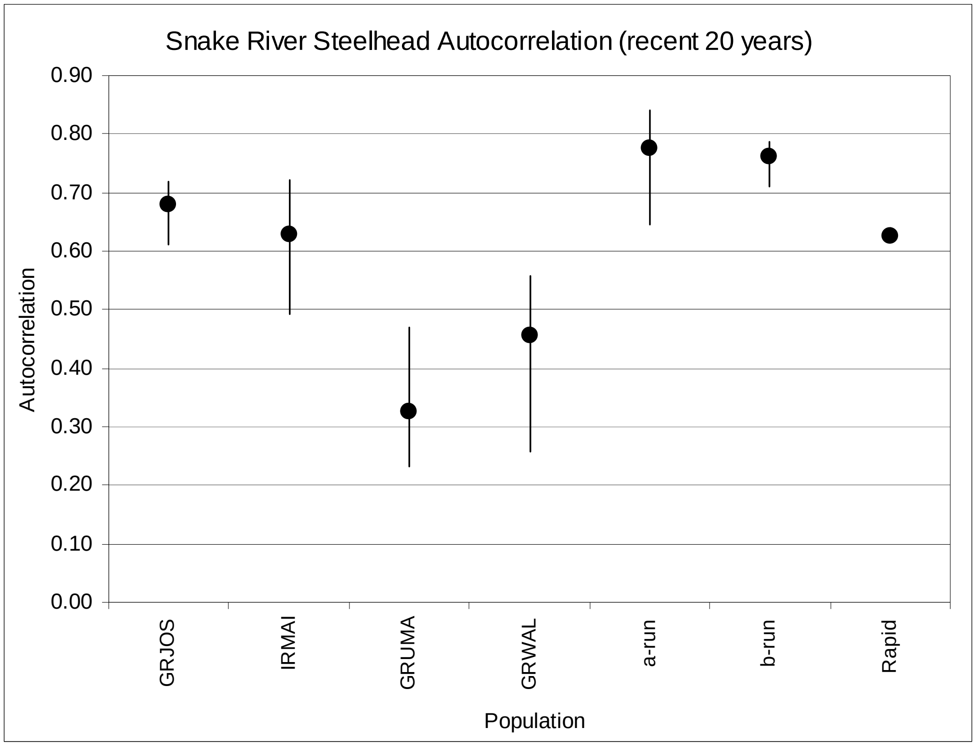


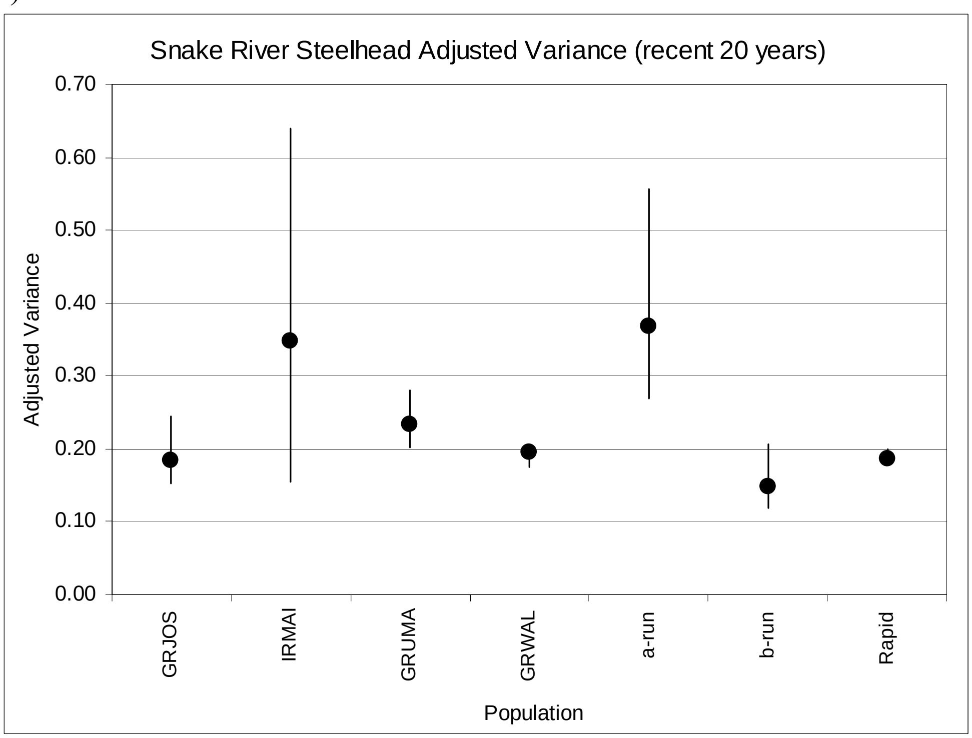
Figure A-11a-c. Population estimates of productivity (geomean brood year spawner to spawner return rates) statistics for the Snake River Steelhead ESU. a) total variance; b) autocorrelation; c) adjusted variance (after accounting for autocorrelation). Bars represent +/- 1 standard error. Filled symbols indicate population data series that met filters described in text.

b)



Snake River Steelhead Autocorrelation (recent 20 years)
Autocorrelation
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
GRJOS
IRMAI
GRUMA
GRWAL
a-run
b-run
Rapid
Population

c)



Snake River Steelhead Adjusted Variance (recent 20 years)
Adjusted Variance
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
GRJOS
IRMAI
GRUMA
GRWAL
a-run
b-run
Rapid
Population

Table A-3. Summary statistics by ESU. Average variance and autocorrelation of residuals from stock/recruit function fits.

		Snake River Spring / Summer Chinook				Upper Columbia Spring Chinook			
	Method	***# of Pops***	***Total Var.***	***Auto***	***Adj. Var.***	***# of Pops***	***Total Var.***	***Auto***	***Adj. Var.***
1	***Original Values***	12	1.18	0.44	0.95	3	0.99	0.68	0.53
2	***Updated Values w original populations***	12	1.29	0.49	0.94	3	1.02	0.68	0.55
3	***Updated Values w all populations***	23	1.52	0.54	1.08	3	1.02	0.68	0.55
4	***no pops w parent esc geomean<50***	18	1.37	0.54	0.97	3	1.02	0.68	0.55
5	***no pops w hatchery > 30%***	18	1.54	0.54	1.09	3	1.02	0.68	0.55
6	***no pops w hatchery OR Stdev > 30%***	17	1.55	0.54	1.10	3	1.02	0.68	0.55
7	***exclude worst fit model***	23	1.43	0.53	1.03	3	0.95	0.68	0.51
8	***4 & 5***	13	1.33	0.55	0.93	3	1.02	0.68	0.55
9	***4, 5 & 7***	13	1.24	0.53	0.89	3	0.95	0.68	0.51

		Snake River Steelhead				Middle Columbia Steelhead			
Number	Method	# of Pops	Total Var.	Auto	Adj. Var.	# of Pops	Total Var.	Auto	Adj. Var.
1	Original Values	2	0.49	0.54	0.35	4	0.44	0.69	0.23
2	Updated Values w original popualtions	2	0.63	0.67	0.34	7	0.54	0.74	0.20
3	Updated Values w all populations	6	0.54	0.61	0.34	13	0.51	0.74	0.23
4	no pops w parent esc geomean<50	6	0.54	0.61	0.34	13	0.51	0.74	0.23
5	no pops w hatchery > 30%	6	0.54	0.61	0.34	12	0.51	0.73	0.24
6	no pops w hatchery OR Stdev > 30%	6	0.54	0.61	0.34	12	0.51	0.73	0.24
7	exclude worst fit model	6	0.39	0.60	0.25	13	0.39	0.75	0.17
8	4 & 5	6	0.54	0.61	0.34	12	0.51	0.73	0.24
9	4, 5 & 7	6	0.39	0.60	0.25	12	0.40	0.74	0.18

		Upper Columbia Steelhead			
Number	Method	# of Pops	Total Var.	Auto	Adj. Var.
1	Original Values	6	0.46	0.64	0.27
2	Updated Values w original popualtions	9	0.56	0.73	0.23
3	Updated Values w all populations	19	0.53	0.70	0.27
4	no pops w parent esc geomean<50	19	0.53	0.70	0.27
5	no pops w hatchery > 30%	18	0.53	0.69	0.28
6	no pops w hatchery OR Stdev > 30%	18	0.53	0.69	0.28
7	exclude worst fit model	19	0.40	0.71	0.2
8	4 & 5	18	0.53	0.69	0.28
9	4, 5 & 7	18	0.38	0.69	0.2

Figure A-12a-d. Snake R. Spring/Summer Chinook ESU viability curves. Variance and autocorrelation parameters used were 0.89 and 0.53, respectively. Age distribution was 0.57 age 4, 0.43 age 5. Minimum abundance thresholds are set for basic, intermediate, and large populations, respectively (Figures a-d).

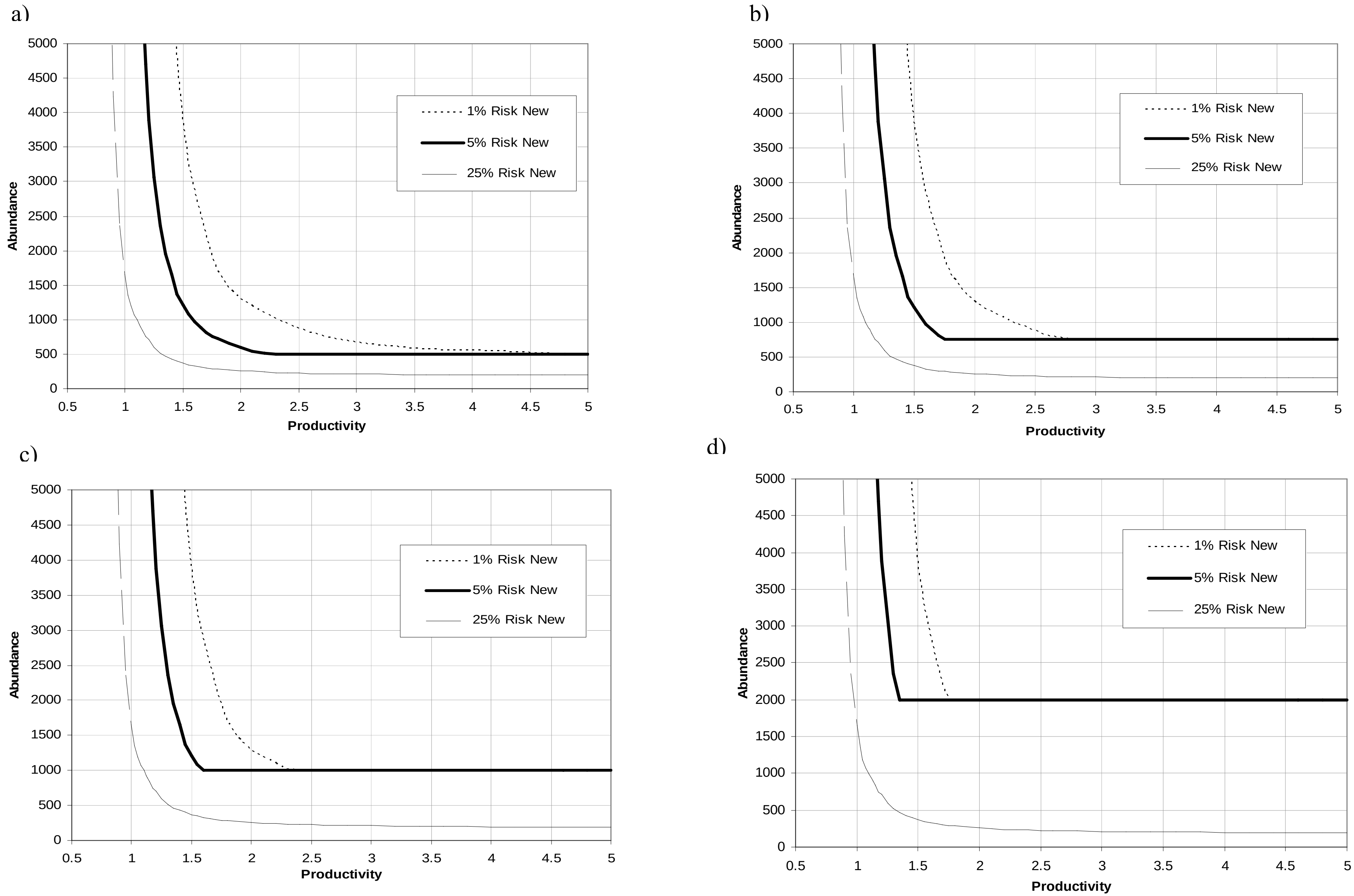
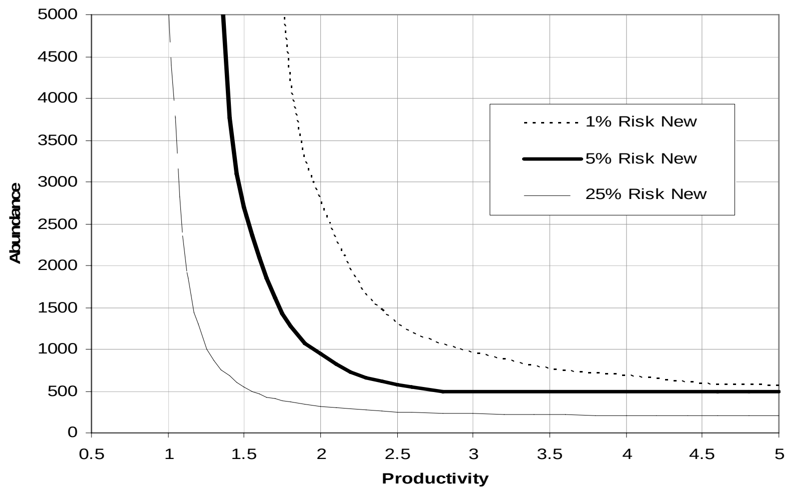
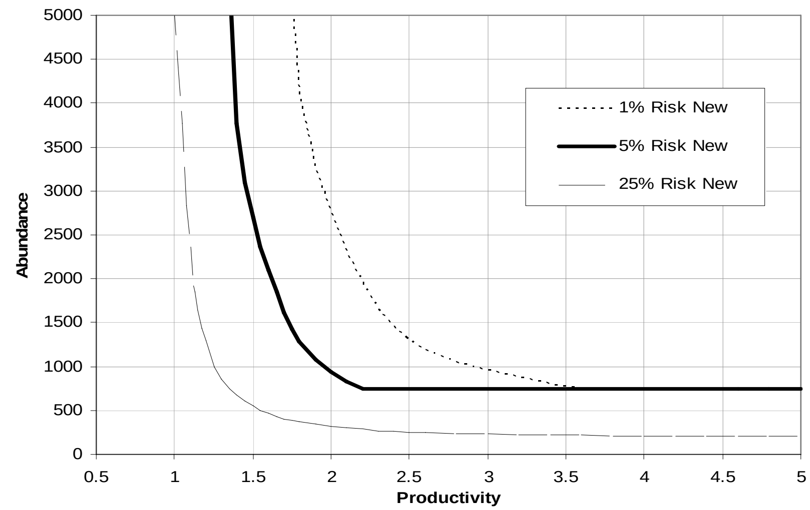


Figure A-13a-d. Upper Columbia Chinook ESU viability curves. Variance and autocorrelation parameters used were 0.51 and 0.68, respectively. Age distribution was 0.60 age 4, 0.40 age 5. Minimum abundance thresholds are set for basic, intermediate, and large populations, respectively (Figures a-d).

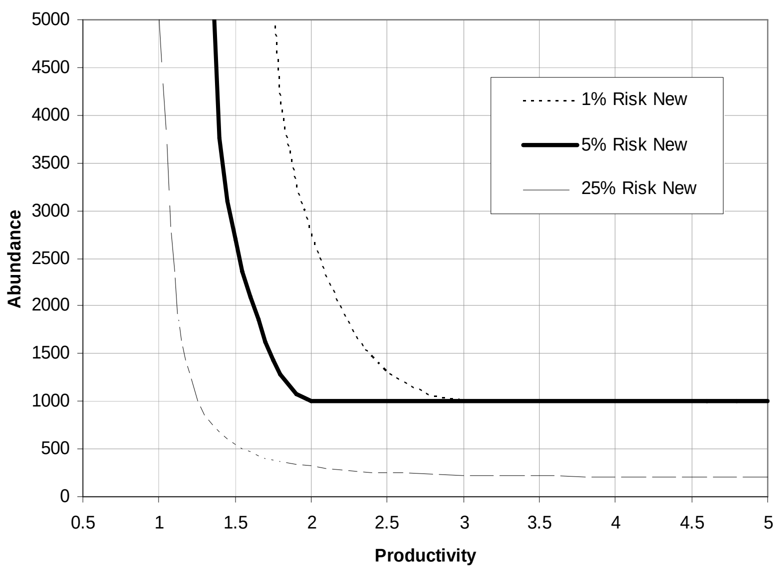
a)



b)



c)



d)

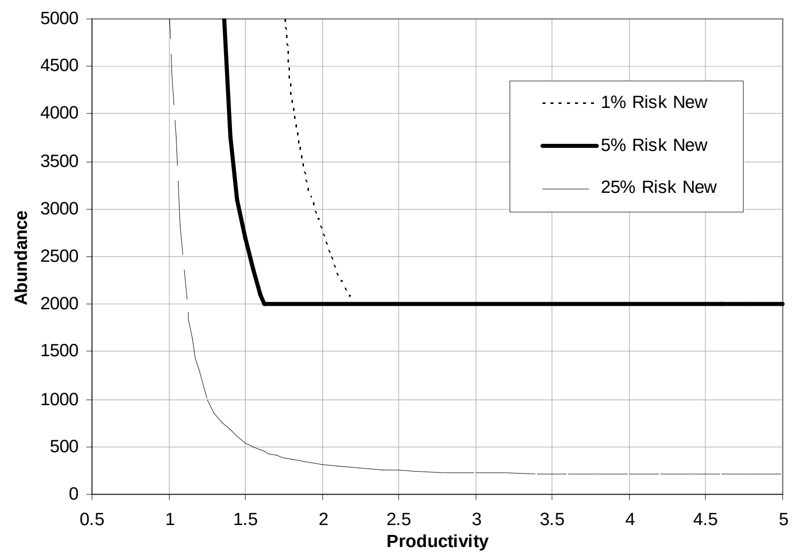
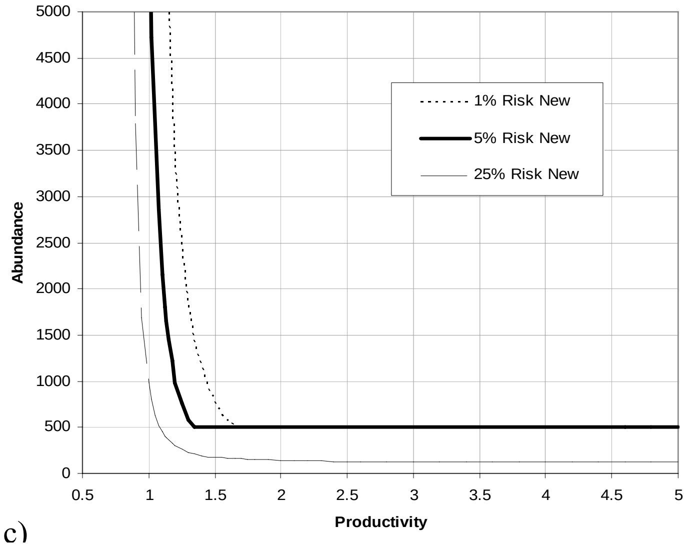
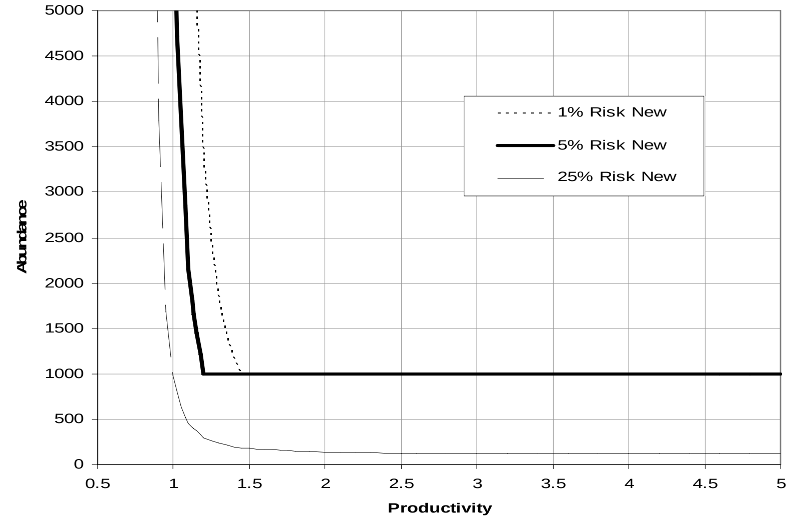


Figure A-14a-d. Upper Columbia Steelhead ESU viability curves. Variance and autocorrelation parameters used were 0.20 and 0.69, respectively. Age distribution was 0.02 age 3, 0.38 age 4, 0.45 age 5, and 0.15 age 6. Minimum abundance thresholds are set for basic, intermediate, and large populations, respectively (Figures a-d).

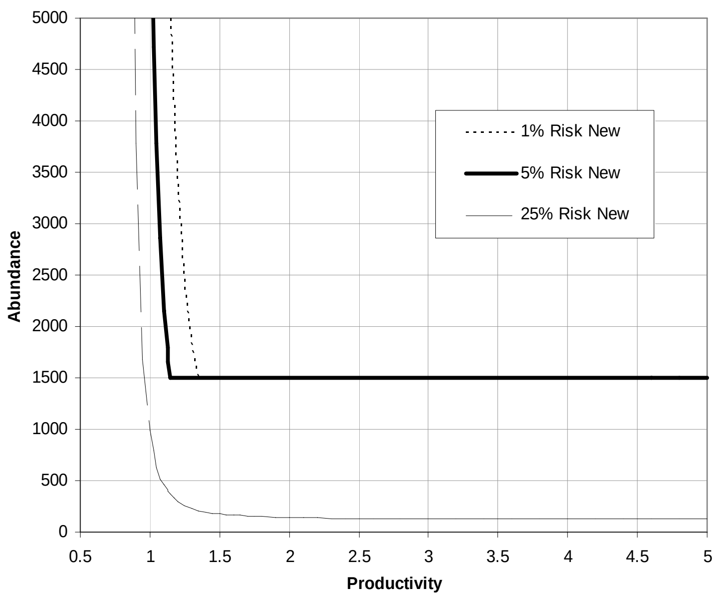
a)



b)



c)



d)

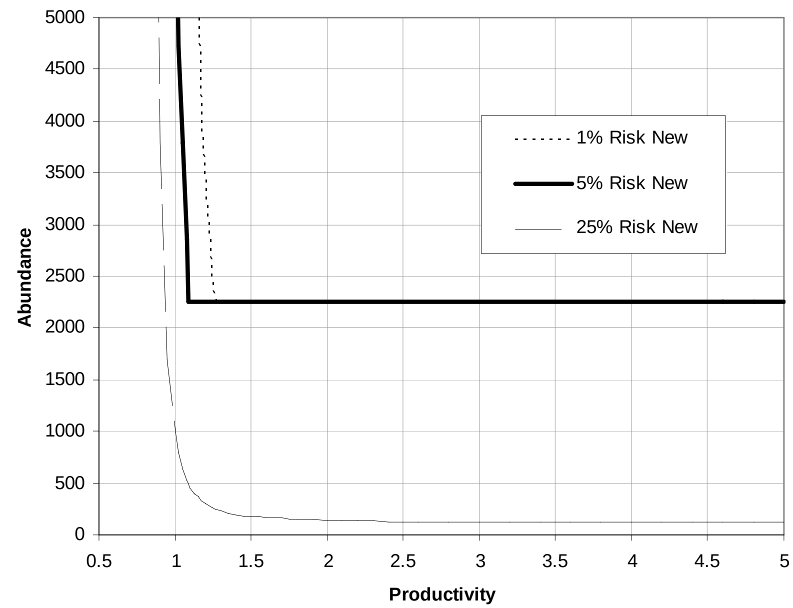


Figure A-15a-d. Middle Columbia Steelhead ESU viability curves. Variance and autocorrelation parameters used were 0.18 and 0.74, respectively. Age distribution was 0.03 age 3, 0.46 age 4, 0.43 age 5, and 0.08 age 6. Minimum abundance thresholds are set for basic, intermediate, and large populations, respectively (Figures a-d).

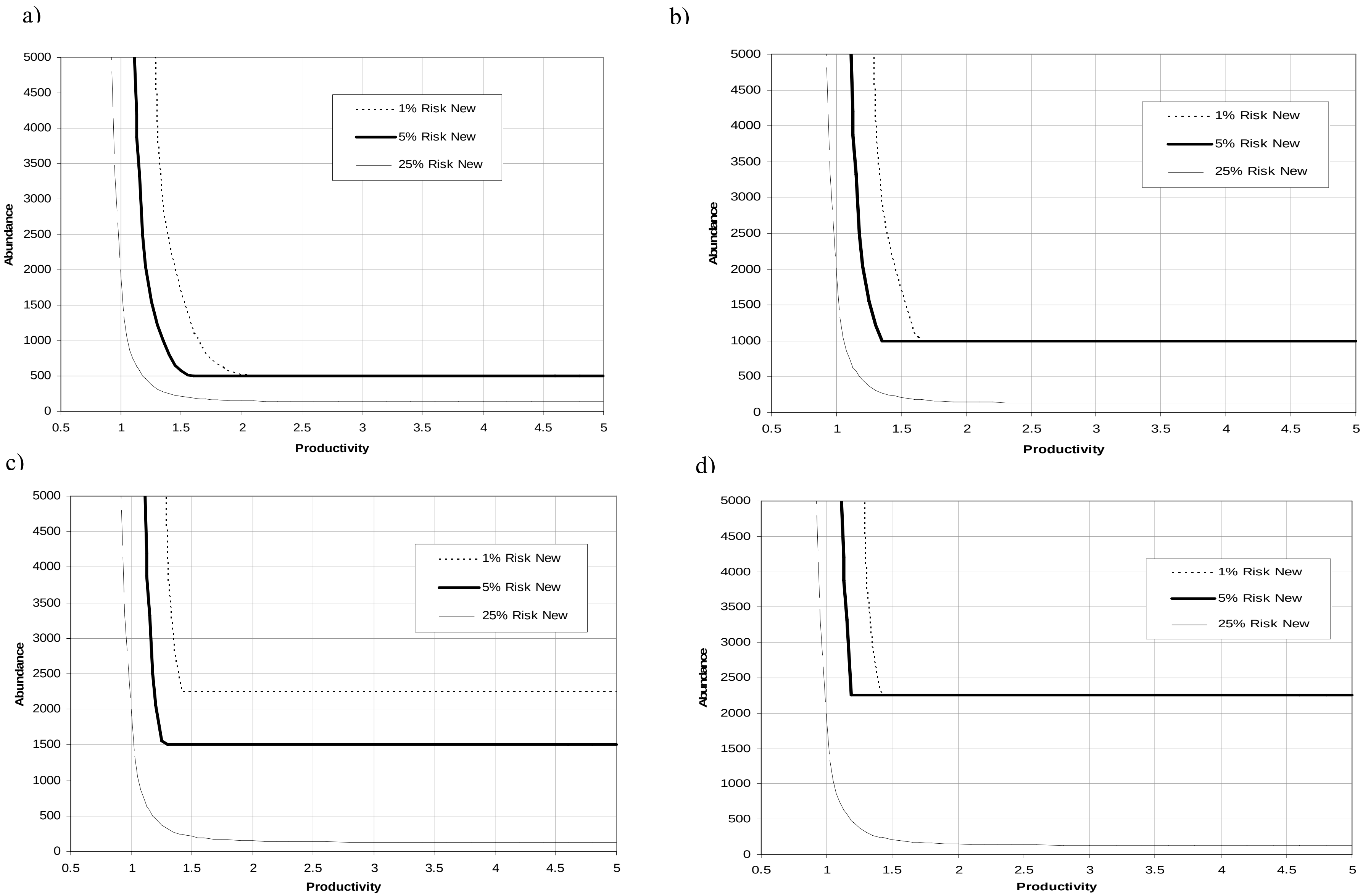
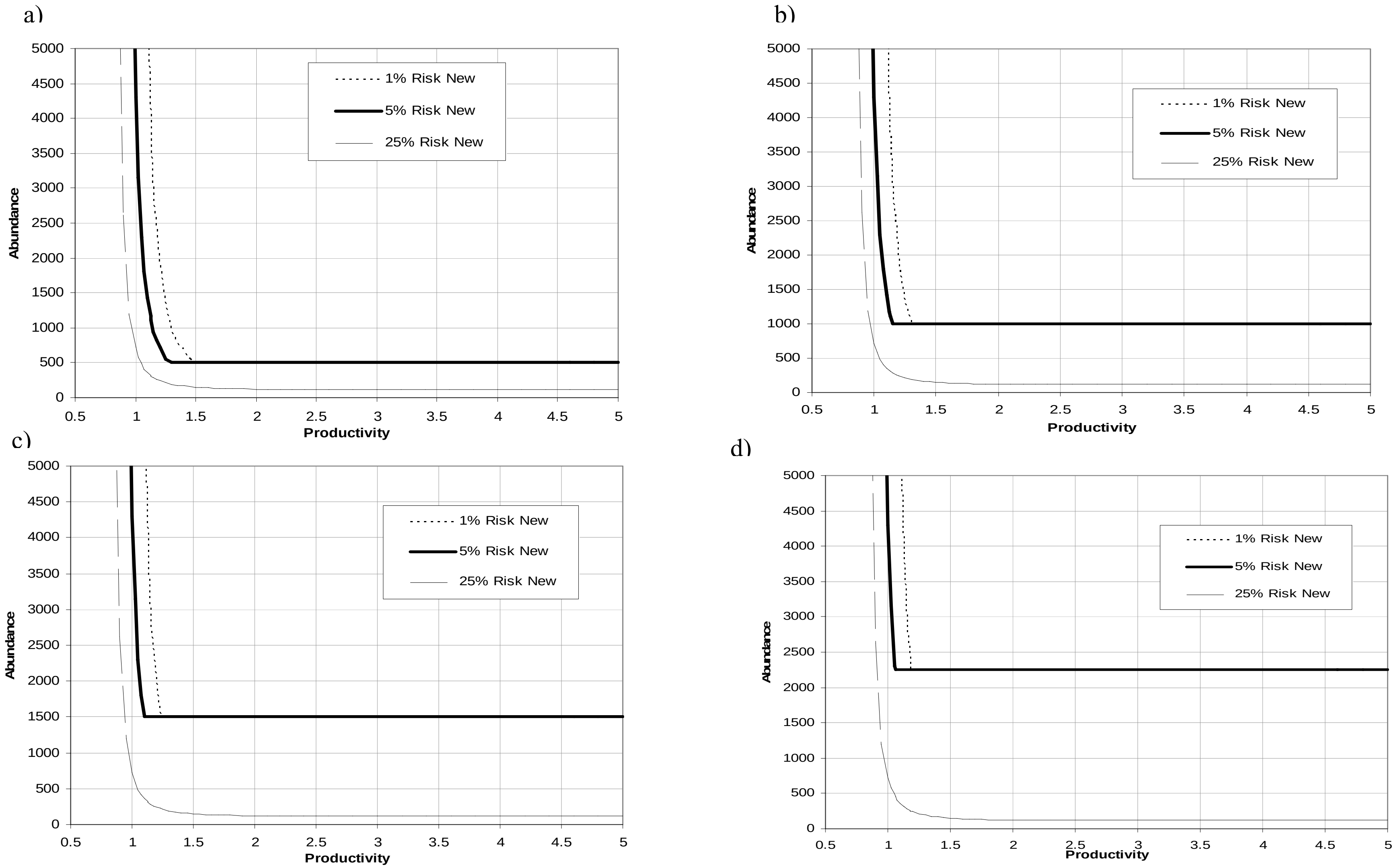


Figure A-16a-d. Snake River Steelhead ESU viability curves. Variance and autocorrelation parameters used were 0.25 and 0.60, respectively. Age distribution was 0.03 age 3, 0.60 age 4, 0.35 age 5, and 0.02 age 6. Minimum abundance thresholds are set for basic, intermediate, and large populations, respectively (Figures a-d).



Fall Chinook ESU

We calculated a viability curve for Snake River fall chinook following the same analytical steps we applied to yearling chinook and steelhead ESUs. We calculated variance and one year lag autocorrelation statistics for reconstructed brood year spawners and natural returns for 1978-2003. We used a grid-search algorithm to develop a set of viability curves for Snake River fall chinook corresponding to projected risk levels of 25%, 5% and 1% at 100 years (Figure A-17).

We established a minimum abundance threshold for fall chinook consistent with the general abundance/productivity objectives summarized in the July 2003 ICTRT Viability draft report. We are recommending a minimum abundance threshold of 3,000 natural origin spawners for the extant Snake River Fall Chinook population. No fewer than 2,500 of those natural origin spawners should be distributed in mainstem Snake River habitat.

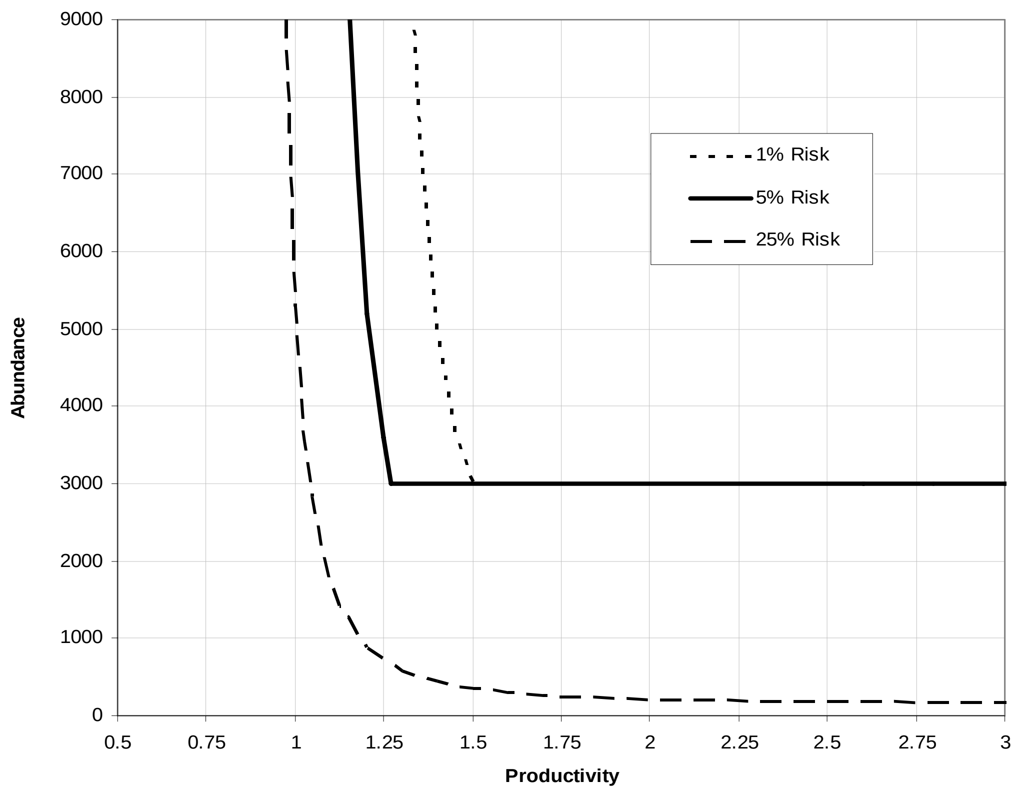


Figure A-17. Viability curves for Snake River Fall chinook. Age structure used was 53% age 3, 43% age 4, and 4% age 5. Adjusted variance (variance unexplained by autocorrelation) and autocorrelation parameters were 0.25 and 0.67, respectively.

The abundance threshold for Snake River fall chinook is based on the Bevan Team recommendation for "…an eight year (approximately 2 generation) geometic mean of at least 2,500 natural origin spawners in the mainstem Snake River annually" (NMFS, 1995). The Bevan Team specifically did not address spawning/rearing areas in the lower mainstems of major tributaries in setting that objective - stating that "…a lack of information precludes setting escapement objectives at this time." It is likely that lower reaches in the Clearwater, Grande Ronde and Tucannon Rivers had the potential to support 500 or more spawners based on physical habitat availability. Fall chinook spawners have been observed in all three areas in recent years (Milks et. al, 2005). Preliminary information from scale sampling and pit tag experiments indicates that natural production of fall chinook in the lower Clearwater may exhibit a complex life history pattern including overwintering in mainstem habitat before outmigrating to the sea the following spring.

Sockeye ESU

Historical sockeye production occurred in at least five Stanley Basin lakes as well as in lake systems associated with Snake River tributaries currently cut off to anadromous access (e.g., Wallowa and Payette Lakes). Current returns of Snake River sockeye are extremely low and are limited to Redfish Lake. In previous ICTRT analyses (McClure et al. 2003, McClure et al. 2005) we have concluded that at least three lakes in the Stanley Lakes Basin historically supported independent sockeye populations (Redfish Lake, Alturas Lake and Stanley Lake).

We do not have a sufficient trend data set specifically for Redfish Lake sockeye to use in generating a viability curve. As a surrogate, we used a data set for Lake Wenatchee sockeye to generate estimates of variance and autocorrelation in return rates (adjusted variance = 0.42, autocorrelation=0.41).

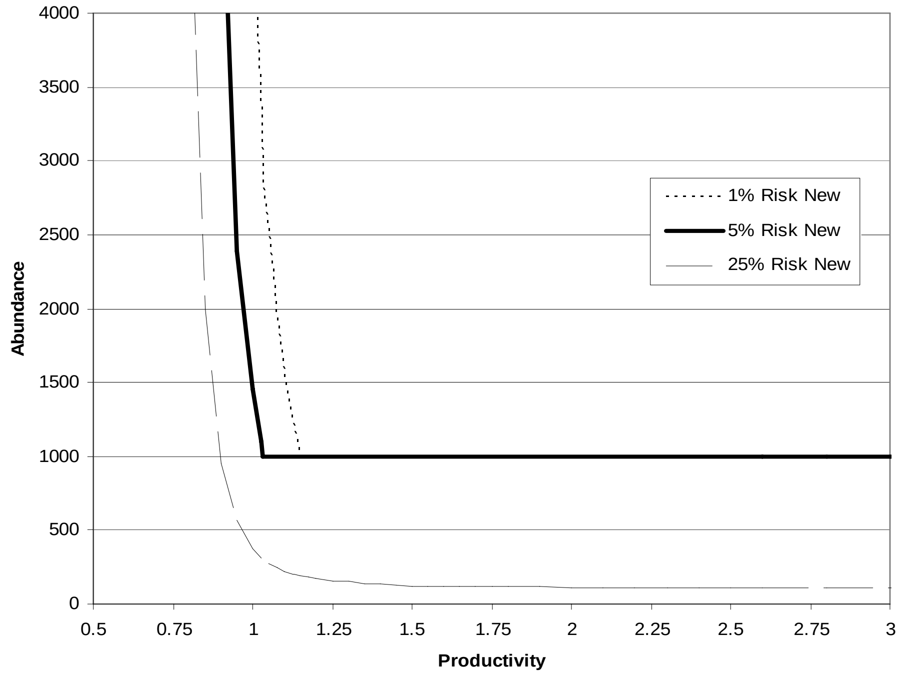
The approach we used to generate a viability curve requires input of a representative adult age structure. Bjornn et al. (1968) identified similarities between Redfish Lake and Wenatchee Lake sockeye runs in age at length and the predominance of 2 year ocean residency in returning adults. We generated an estimate of average age structure for Redfish Lake sockeye using smolt age sampling data summarized in Bjornn et al. (1968) as a starting point. Redfish Lake sockeye smolts outmigrated after one or two years residency in freshwater. The proportions varied considerably across brood years, The median proportion age 1 migrants for the 1954 to 1963 year classes was 0.60. Information cited in Bjornn et al. (1968) indicates that almost all returning adults had spent 2 years at sea. Based on these estimates, we assumed that the average age composition of returning adult Redfish Lake sockeye was 60% 4 year olds and 40% 5 year olds.

We generated two sets of curves for application to potential Stanley Lake Basin sockeye populations (Figure A-18). We developed relative population size category designations for Columbia Basin lake systems based on relative surface areas (Appendix B). The Stanley Basin Lakes are relatively small compared to other lake systems that historically supported sockeye production in the Columbia Basin. Stanley Lake is assigned to the smallest size category along with Pettit and Yellowbelly Lakes. Redfish Lake and Alturas Lake fall into the next size category – Intermediate. We adapted the recovery abundance levels recommended by the Snake River Recovery Team (Bevan, et al. 1994) as minimum abundance thresholds. We set the minimum spawning abundance threshold at 1,000 for the Redfish and Alturas Lake populations (intermediate category), and at 500 for populations in the smallest historical size category (e.g., Stanley Lake).

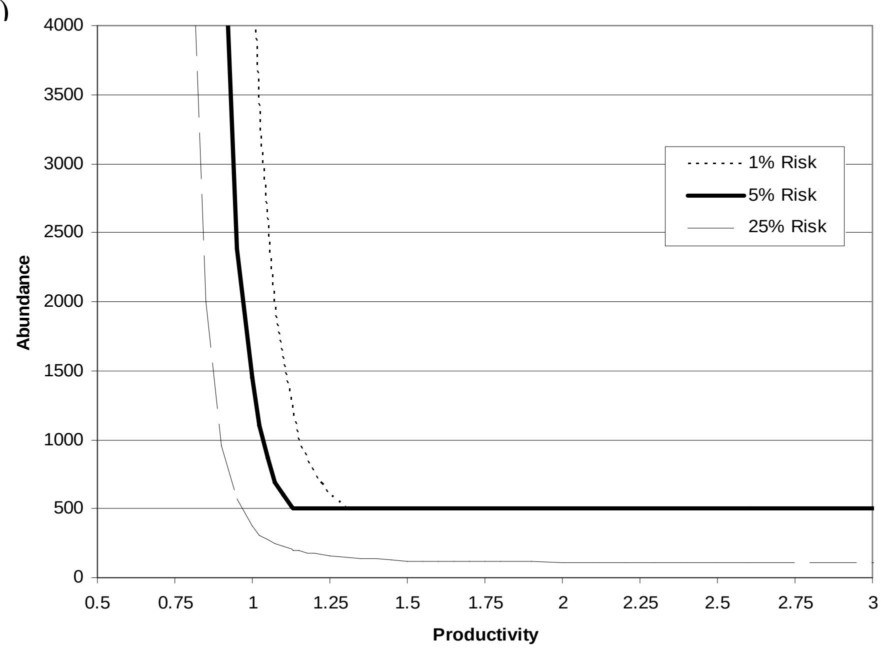
These estimates should be viewed as interim long-term abundance/productivity objectives for Stanley Basin sockeye populations. Returns of Snake River sockeye have been at extremely low levels for a considerable period of time. Initial efforts aimed at recovery will likely put a high priority on increasing survival of juvenile outmigrants and adult returns to levels that will allow for rebuilding. Information on juvenile productivity and on specific year to year variations in Redfish Lake brood year return rates gathered during the initial phase of recovery efforts should allow for future refinements of the interim ICTRT Snake River sockeye abundance and productivity criteria.

Figure A-18a-b. Viability curves for application to Snake River sockeye lake populations. A) Redfish Lake and Alturas Lake (Intermediate); B) small lake populations (Stanley Lake). Age structure used was 60% age 4 and 40% age 5 adult returns. Adjusted variance (variance unexplained by autocorrelation) and autocorrelation parameters (derived from Lake Wenatchee data) were 0.42 and 0.41, respectively.

a)



b)



Updating Viability Curves

The ICTRT developed a set of viability curves based analyses of trend data sets available (or applicable) for each ESU as of December, 2005. We recommend that these curves be periodically reviewed and updated as appropriate. At a minimum, additional return year data will become available for each series. Techniques for estimating escapements for populations may be improved, leading to revisions in the estimates used in generating the viability curves. Additional data series may become available. The ICTRT recommends that viability curves should be comprehensively reviewed and updated every 5 years, in phase with periodic population status updates. The choice of a five year interval reflects a balance between ensuring that recovery targets are based on updated information and avoiding frequent, minor changes to criteria resulting from yearly updates. We recommend using a test to ensure that updates leading to relatively substantial changes in viability curves are incorporated, while minimizing the need to update all analyses dependent upon viability curves in response to relatively minor shifts.

The viability curves for Interior Columbia ESUs reflect specific estimates of variance and autocorrelation in return rates. Estimates of these two parameters can be updated as escapement estimates become available for each additional year, or as a result of revisions to run reconstruction methods. We developed the following test to highlight when changes in those estimates are sufficiently large to warrant updating viability curves used in recovery planning.

1) Generate an updated version of the 5% viability curve for the Basic size population grouping of the ESU under consideration.

2) Compare the resulting curve to the current (without data updates) versions of the 1%, 5% and 25% risk curves for the ESU at abundance levels between 500 and 1000.

    a. To facilitate the comparison, calculate intermediate risk curves for intermediate levels (3%, 15%) using for the current (without data updates) data.

3) Adopt the updated viability curve parameters IF:

    a. The updated version of the 5% curve exceeds the curve associated with a 3% risk of extinction (previous data set), or

    b. The 5% curve falls below the curve associated with a 15% risk (previous data set)

Sensitivity Analyses
Viability Curve Input Parameters

The input parameters driving the form of ESU specific viability curves are each subject to substantial process and measurement uncertainties. We evaluated the sensitivity of viability curves to variations in the input values for variance and autocorrelation in intrinsic productivity and in average age structure. We used the average values calculated from Snake River spring/summer chinook population data sets as a baseline for the sensitivity assessment. We structured the sensitivity analysis to allow for comparisons of the impact of proportional variations across the three input parameters. We generated a range of values for each input parameter using a common set of proportional multipliers (Table A-4).

We evaluated the effects of sequentially varying each of the three input parameters on the viability curves. We generated a set of viability curve parameters corresponding to each of the three inputs. In any given set, the remaining two input parameters were maintained at the baseline level.

Table A-4. Range of input parameters used in viability curve sensitivity analyses.

Proportion of Input Value	Viability Curve Parameter		
	Total Variance (geomean productivity)	Autocorrelation (geomean productivity)	Age Structure (4 yr old proportion)
2.00 x	2.48	--	--
1.50 x	1.86	0.80	.85
1.25 x	1.55	0.65	.71
1.00 x	**1.24**	**0.53**	**.57**
0.75 x	0.93	0.40	.42
0.50 x	0.62	0.27	.28
0.25 x	0.31	0.14	.14

The QET and RFT were held at baseline levels for the variance, autocorrelation and age structure sensitivity runs. In a separate analysis, we evaluated the impact on viability curves of incorporating different values for QET and for RFT.

We used consistent metrics for contrasting the results of the sensitivity runs to facilitate comparisons. We expressed the results of the individual parameter analyses in terms of the minimum productivity associated with threshold abundance levels for the four size categories of spring/summer chinook populations (i.e., 500, 750, 1000 and 2000).

Variance and Autocorrelation

Projected viability curves are particularly sensitive to input parameters for variance and autocorrelation in productivity (spawner to spawner return rate).

The effect of total variance on the minimum productivity at threshold abundance levels is most pronounced for the basic population category (Table A-5a). Holding all other input parameters at their average values and setting the total variance at 0.75 and 1.25 times the average level used in generating spring/summer chinook viability curves changes the minimum productivity at threshold abundance by -24% and +47%, respectively. The relative change at higher abundance levels is dampened, but follows the same pattern.

Proportionally varying the level of autocorrelation input (holding other input variables constant) also had a substantial effect on the projected viability curve (Table A-5b). The average autocorrelation for the Snake River Spring/Summer Chinook ESU populations was 0.53. Increasing the input value for autocorrelation by 25% or more resulted in substantial increases in the required productivity at threshold abundance levels.

Table A-5a. Estimated productivities as a function of **total variance in productivity** (spawner to spawner return rates). Results at Snake River Spring/Summer Chinook ESU average total variance are in bold type. Results are presented as productivities corresponding to minimum equilibrium escapement levels (5% risk) by population size category (basic, intermediate, large and extra large). All other viability curve input parameters are held at recent geomeans for Snake River spring summer ESU populations.

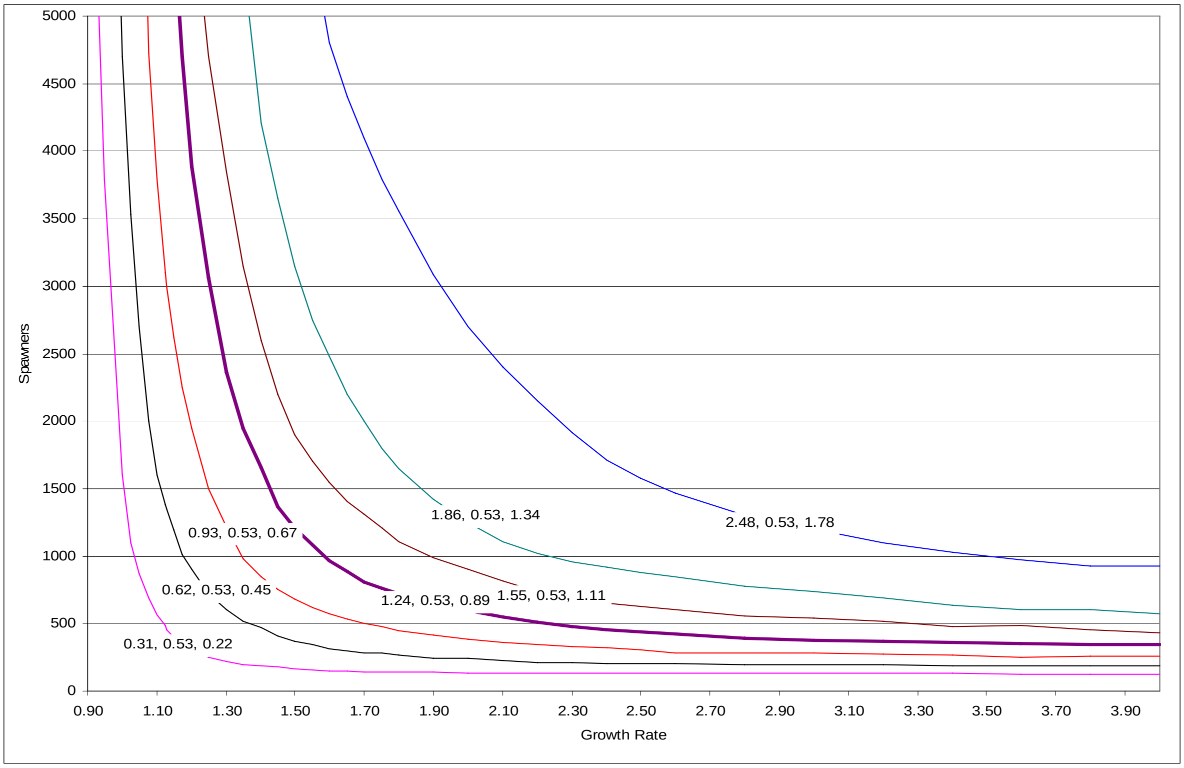
Total Variance (spawner to spawner return rate)	Minimum Population Size			
	500	750	1000	2000
0.31	1.11	1.08	1.04	0.98
0.62	1.34	1.25	1.17	1.08
0.93	1.69	1.44	1.38	1.19
1.24	**2.21**	**1.76**	**1.56**	**1.34**
1.55	3.25	2.22	1.82	1.48
1.86	5.60	2.88	2.22	1.70
2.48	6.00+	5.00+	3.42	2.22

Table A-5b. Estimated productivities as a function of **autocorrelation in productivity** (spawner to spawner return rates). Results at Snake River Spring/Summer Chinook ESU average total variance are in bold type.

Autocorrelation (Spawner to spawner return rate)	Minimum Population Size			
	500	750	1000	2000
0	0.95	0.88	0.85	n/a
0.13	1.06	0.98	0.93	0.85
0.27	1.25	1.13	1.07	0.96
0.53	**2.21**	**1.76**	**1.56**	**1.34**
0.66	4.10	2.60	2.25	1.78
0.80	5.00+	5.00+	4.30	3.20

Figure A-19a-b. Sensitivity of Snake River Spring/Summer Chinook viability curve to a) a range of total variance input values above and below the ESU average (1.24 total variance, 0.89 after adjustment for autocorrelation, autocorrelation fixed at ESU average level of 0.53); and b) autocorrelation input values.

a)



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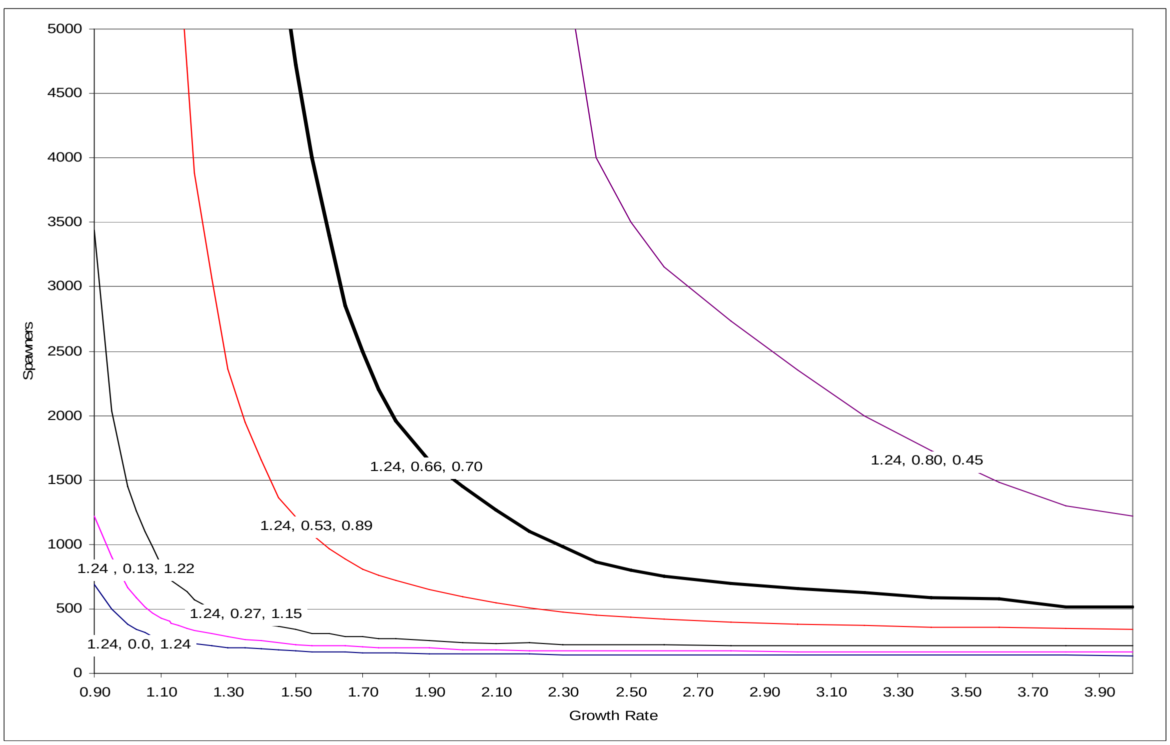


Age structure

Adult spawning returns for Interior Columbia stream type chinook populations are predominated by 4 and 5 year old fish. In many years a relatively small component of 3 year old returns are present, virtually all of these fish are males. A small percentage of mature adults return at age 6. For the purposes of this analysis we included those fish as age 5 returns. The viability curves derived for Snake River Spring/Summer Chinook population categories incorporate an average age composition for the ESU (0.57 age 4, 0.43 age 5 returns). We systematically varied age composition (Table A-4) and evaluated the sensitivity of projected viability curves, holding other input parameters at the recent average values used in constructing the viability curves for this ESU presented in the ICTRT viability report.

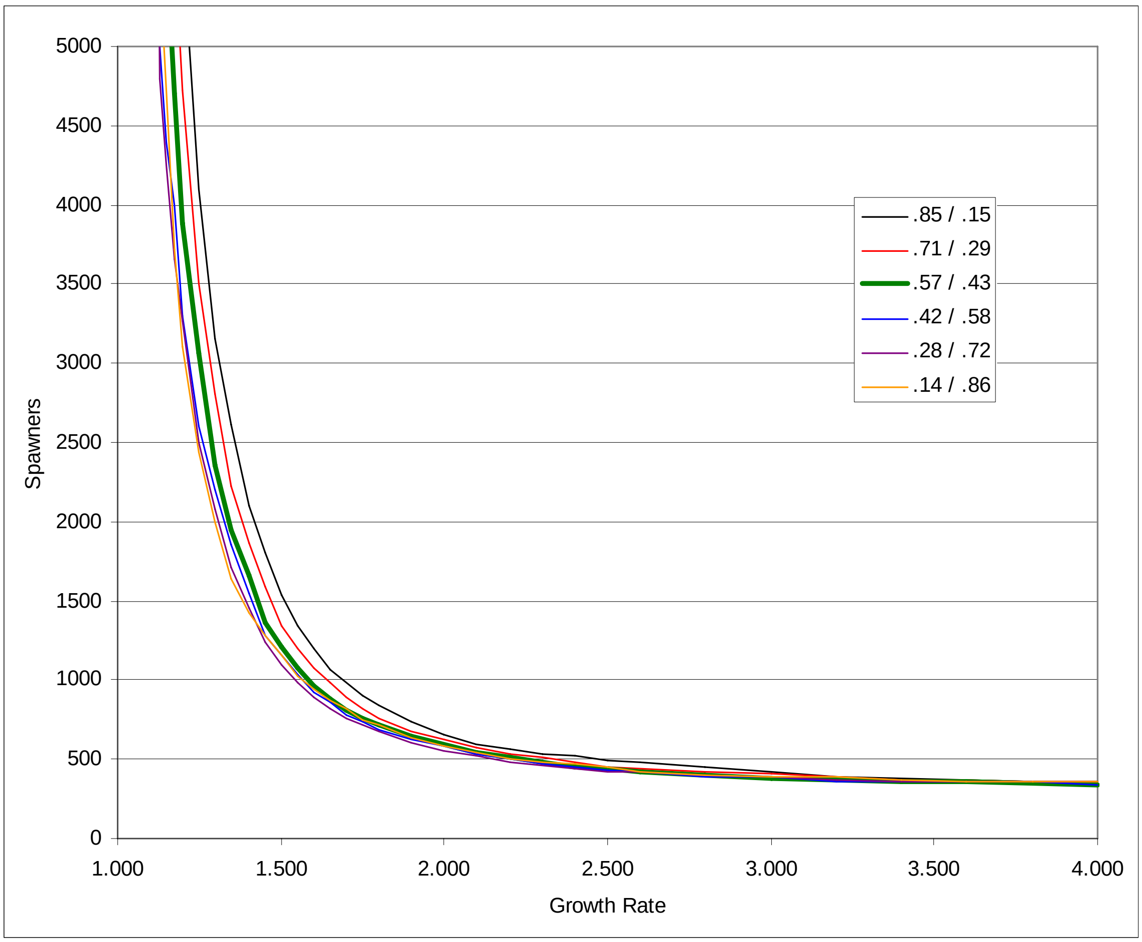


Figure A-20. Sensitivity of a Snake River Spring/Summer Chinook 5% risk viability curve to a range of age structures above and below the ESU average (0.57 age 4; 0.43 age 5). Total variance and autocorrelation were maintained at ESU average levels of 1.24 and 0.53, respectively. A QET of 50 adult spawners per year for four years was used.

Variations on the average age composition resulted in relatively small changes to projected viability curves (Figure A-20, Table A-6). The relative change in the productivity associated with minimum abundance was greatest for the basic population size category. Reducing the proportion 4 year olds by half decreased the required productivity by approximately 10%, while increasing the proportion by 1.5 resulted in a relative increase of approximately 10% . Changes for other size categories were generally lower (+9% to -4% at the limits of the range in input values).

Table A-6. Estimated productivities as a function of **average age structure** (results at ESU average age structure in bold type). Results are presented as productivities corresponding to minimum equilibrium escapement levels by population size category (basic, intermediate, large and extra large). All other viability curve input parameters are held at recent geomeans for Snake River Spring/Summer Chinook ESU populations.

Age Structure (Prop. 4/Prop. 5 year old spawners)	Minimum Population Size			
	500	750	1000	2000
0.85 /0.15	2.45	1.78	1.72	1.43
0.71 / 0.29	2.29	1.77	1.68	1.39
0.57 / 0.43	**2.21**	**1.76**	**1.56**	**1.34**
0.42 / 0.58	2.20	1.73	1.54	1.34
0.28 / 0.72	2.16	1.71	1.53	1.31
0.14 / 0.86	2.13	1.70	1.51	1.30

Quasi-Extinction Threshold (QET)

The ICTRT viability curves were generated using a QET value of 50 spawners per year for a four year period. We evaluated the sensitivity of the projected viability curves to a range of QET input values. The range of QET values tested included an alternative corresponding to explicit extinction (less than 2 spawners per year), multiples of the 50 spawners per year value used by the ICTRT, and three larger values (150, 200 and 250 spawners per year) corresponding to thresholds applied to populations classified as Medium and Large in LC-WTRT analyses for application to Lower Columbia ESUs (LCWTRT, 2006 viability draft ref).

We generated viability curves (5% risk over 100 years) for each QET value (Figure A-21). To facilitate comparisons, we expressed the results as minimum productivities associated with meeting threshold population size values for Interior Columbia basin Snake River Spring/Summer Chinook populations (Table A-7).

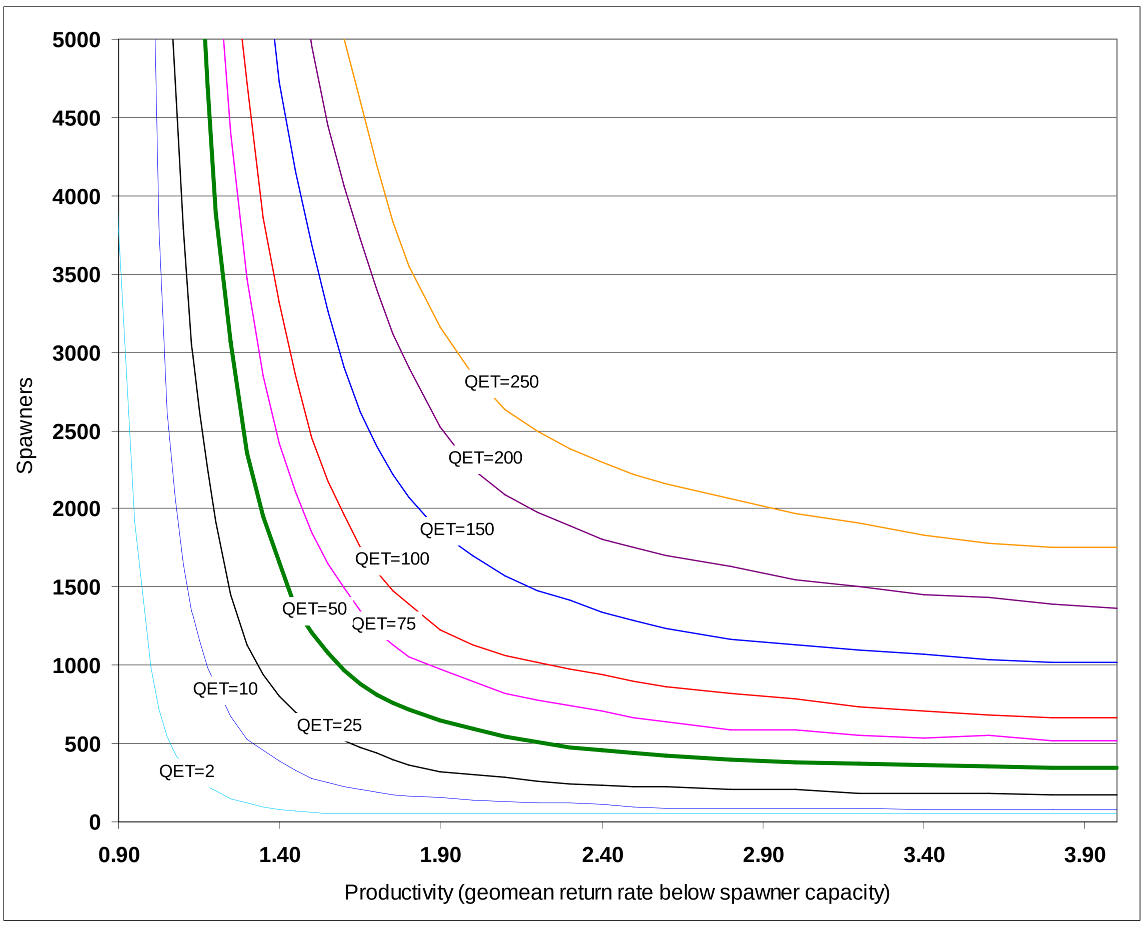


Figure A-21. Sensitivity of Snake River Spring/Summer Chinook viability curve to a range of QET values above and below the level of 50 spawners/year adopted by the ICTRT (1.24 total variance, 0.89 after adjustment for autocorrelation). The RFT was set at 10 in the model runs for QET values of 10 or greater. The RFT was set at 2 for runs in which the QET was 2.

Table A-7. Sensitivity analysis of **QET input values**. Estimated productivities at minimum equilibrium escapement levels corresponding to alternative population size classes. QET values greater than 100 were included to facilitate comparison to LC-WTRT analyses for larger population categories. In this analysis, the reproductive failure threshold (RFT) was set to 10 spawners except for the QET of 2 (RFT was also set to 2 in this case).

QET Threshold Escapement	Minimum Population Size			
	500	750	1000	2000
2	1.05	1.03	1.00	0.95
10	1.36	1.22	1.18	1.08
25	1.60	1.42	1.34	1.19
50	**2.21**	**1.76**	**1.56**	**1.34**
100	10.00+	3.50	2.27	1.58
150	10.00+	10.00+	4.20	1.87
200	10.00+	10.00+	10.00+	2.20
250	10.00+	10.00+	10.00+	2.90

The productivities required to meet or exceed the viability curves at minimum average population abundance levels were substantially affected by the choice of a QET value. Increasing the QET value from 50 to 100 roughly doubled the required productivity at threshold abundance levels for the two smallest population size categories. The productivities at threshold abundance levels were increased by approximately 45% for the large category and by 18% for the extra large population size category.

Setting the QET at 25 spawners per year reduced productivities associated with population size category minimum abundance levels by 28% (basic) to 11%(very large).

Setting the QET at 2 fish reduced the projected average productivities at population size category abundance thresholds by 29% to 52% relative to requirements associated with the QET of 50 spawners per year. The relative reductions in required productivity are greatest for populations within the basic size grouping.

We conducted two additional analyses of the sensitivity of model risk projections to the choice of a QFT value. One set of tests evaluated the impact of the choice of a QET input on the proportion of relatively low escapements in projected model runs. The second test evaluated the relative impact of incorporating 'the wrong' QET value.

A major rationale in setting the QET at 50 spawners per year in establishing viability curves for Interior Columbia ESU populations was the uncertainty associated with productivities at escapements that were below levels in the historical record. Model runs incorporating lower QETs would be expected to project higher proportions of annual escapements below 50 spawners, even when the productivity and abundance levels incorporated into the runs reflect projected extinction risk of 5% or less. We compared model runs incorporating the range of QET values summarized in Table A-6 to evaluate the impact of QET on the expected proportion of relatively low escapements. The RFT was set at 10 fish for all of the QET values except the lowest value (QET = 2). In that case, the RFT was also set at 2 spawners. Each of the model runs incorporated input parameters corresponding to a 5% risk of extinction in 100 years for the particular QET being tested in the run. We calculated the expected proportion of annual spawning escapements at relatively low escapement levels as a function of QET (Table A-8) .

The number of 100 year simulation runs out of 1000 with a relatively high proportion of escapements below 50 spawners increased as QET was decreased. The proportion of relatively low escapements increased substantially when the QET was lowered from 10 to 2 spawners.

Table A-8. Comparison of the incidence of projected annual spawning escapements below 50 spawners per year as a function of QET. Equilibrium abundance was set at 500 spawners. Productivity was set at the level corresponding to a projected risk of 5% over 100 years. RFT used in model runs in parentheses.

Assigned QET (RFT)	Number of annual spawning escapements less than 50 (in 100 year model runs)		
	10 or more	20 or more	30 or more
2 (2)	46.6%	27.7%	19.4%
10 (10)	20.6	8.4	5.3
25 (10)	12.1%	3.8%	2.2%
50 (10)	1.7%	0.1%	0.0%

We evaluated the potential effects of setting the QET value at a particular level when the 'true' QET is at a different value. We ran these model runs with an equilibrium population abundance of 500 spawners. We ran a set of model projections for each combination of assumed and underlying actual QET values. For each combination, the productivity associated with a 5% risk for the assumed QET was used as input. We ran the model with the actual QET to determine the projected risk associated with the input productivity. The results are summarized in Table A-9. For example, the projected risk of extinction in 100 years if the actual QET value is 50 but the assumed value is 2 would be 47%. Conversely, if the actual QET value is 2 and the assumed QET is 10, the projected 100 year risk is 0.2% (Table A-9).

Table A-9. Comparison of projected risks across productivities associated with 5% risk at for a basic population with an equilibrium population size of 500. Rows: assigned QET (productivity in parentheses). Columns correspond to actual QET incorporated into model runs. Entries are the projected extinction risk for the combination of assigned and modeled QET. Reproductive failure threshold (RFT) was set to 10 spawners except when QET = 2 (RFT was set to 2 in these cases).

Assigned QET (prod @ threshold)	Effective (Actual) QET			
	2	10	25	50
2 (1.05)	**5%**	19%	30%	47%
10 (1.36)	0.2%	**5%**	11%	22%
25 (1.60)	0.1%	2%	**5%**	14%
50 (2.21)	0.0%	0.2%	1%	**5%**

Reproductive Failure Threshold (RFT)

The stochastic population viability model used to generate viability curves incorporates a reproductive failure threshold (RFT). For each particular set of input parameters being tested, the model generates a minimum of 1,000 simulations of population performance projected over 100 or more years. Each of the 100 year simulation runs is structured as a series of annual time steps, using the age structure input values to distribute production from a particular brood year across future return years. If spawning escapement in any particular year falls below the RFT value, production from that brood year is set to zero. As a result, there would be no contributions from that particular brood year to future return years. We evaluated four alternative RFT values ranging from 2 to 50 spawners, holding other input values at the levels used in generating the viability curves (table A-10).

Table A-10. Sensitivity analysis of **RFT input values**. Estimated productivities needed to achieve 5% risk at minimum equilibrium escapement levels corresponding to alternative population size classes. The QET was held at 50 spawners for four consecutive years in all runs.

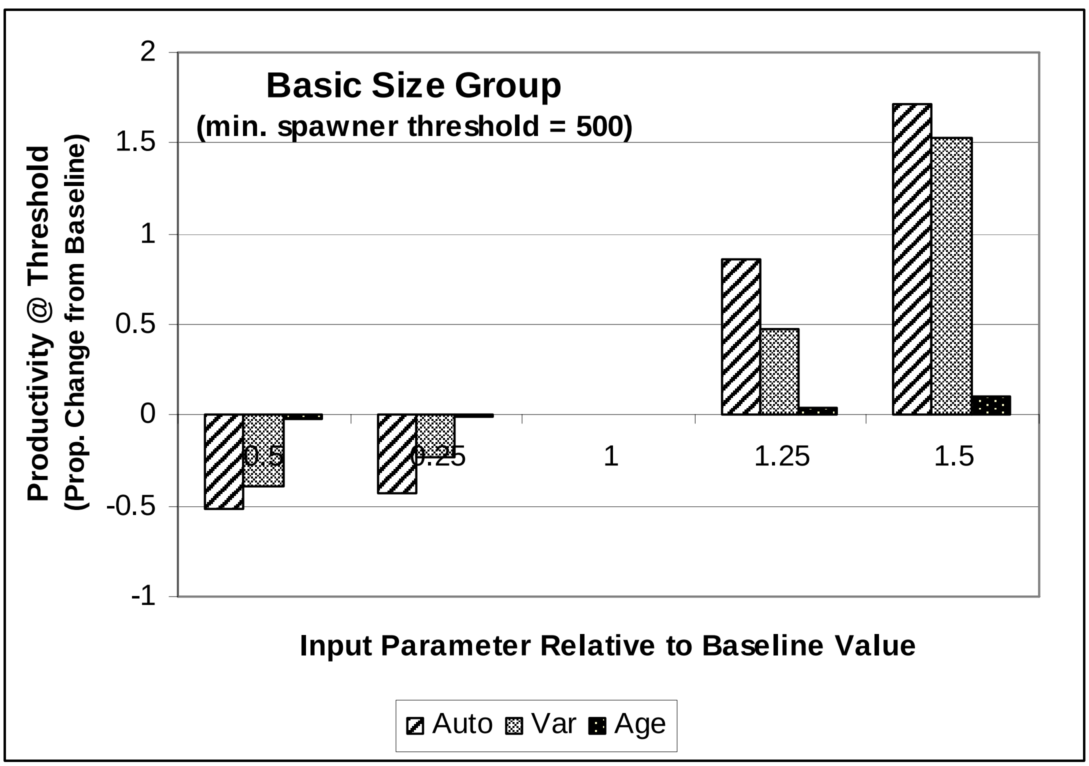
RFT Escapement	Minimum Population Size			
	500	750	1000	2000
2	2.10	1.73	1.54	1.32
10	**2.21**	**1.76**	**1.56**	**1.34**
25	2.28	1.79	1.60	1.36
50	2.43	1.93	1.69	1.41

Relative Sensitivity

We compared the relative sensitivity of projected viability curves to proportional changes in the three population specific input factors. We used the estimated productivities at equilibrium spawning level (500 and 1,000) corresponding to a projected risk level of 5% extinction in 100 years as a standard index of the viability curves. The projected curves were most sensitive to alternative values of autocorrelation in annual productivities (Figure 22). Variations in the input value for total productivity also generated substantial changes in the relative position of the viability curve. Variations in average age structure did not substantially impact the position of the curve in these examples. Viability curves with a minimum abundance threshold for application to relatively small populations (i.e., the Basic size category) were more sensitive to modest variations in the input parameters for autocorrelation and total variance than curves with a Large population size threshold (1,000). Increasing the autocorrelation input value above 0.80 resulted in a substantial increase in the projected productivities for the large size category as well.

Figure A-22a-b. Relative effects of proportional variations in population input parameters on estimated productivity associated with a projected 5% risk of extinction at equilibrium population size of 500 spawners. Initial input values were geomean estimates for Snake River spring/summer chinook populations. Each parameter was varied from by a standard set of proportions (see Table A-4).

a)



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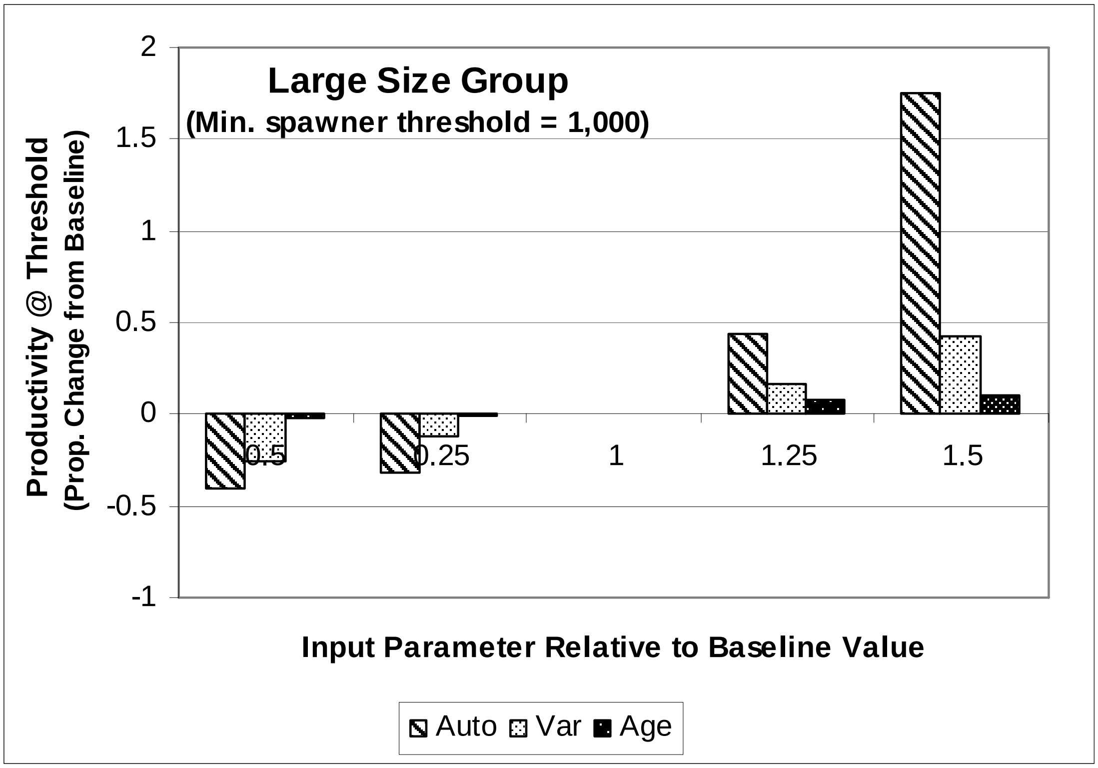

Large Size Group
(Min. spawner threshold = 1,000)
Productivity @ Threshold
(Prop. Change from Baseline)
2
1.5
1
0.5
0
-0.5
-1
0.5
0.25
1
1.25
1.5
Input Parameter Relative to Baseline Value
Auto
Var
Age

Literature Cited

Bevan, D., J. Harville, P. Bergman, T. Bjornn, J. Crutchfield, P. Klingeman and J. Litchfield. 1994. Snake River Recovery Team: final recommendations to National Marine Fisheries Service. May 1994.

Burgner, R.L. 1991. The life history of sockeye salmon (Oncorhynchus nerka). In: C. Groot and L. Margolis (eds.), Life history of Pacific salmon. p. 3 -117. Univ. of British Columbia Press. Vancouver, B.C.

Bjornn, T.C., D.R. Craddock, D.R. Corley. 1968 chk Migration and survival of Redfish Lake, Idaho sockeye salmon, *Oncorhynchus nerka*. Trans. Amer. Fish. Soc. :360-373.

Holmes, E.J. 2001. Estimating risks in declining populations with poor data. Proc. Natl. Acad. Sci. 98(9):5072-5077.

Ginzberg, L.R., L.B. Slobodkin, K. Johnson and A.G. Bindman. 1982. Quasi-extinction probabilities as a measure of impact on population growth. Risk Analysis 2:171-181.

McElhany, P., T. Backman, C. Busack, S. Heppell, S. Kolmes, A. Maule, J. Myers, D. Rawding, D. Shively, A. Steel, C. Steward & T. Whitesel. 2003. Draft Willamette/Lower Columbia Pacific Salmonid Viability Criteria. March, 2003

McElhany, P. Ruckelshaus, M.H., Ford, M.J., Wainwright, T.C. and Bjorkstedt, E.P. 2000. Viable salmonid populations and the recovery of evolutionarily significant units. Technical Memorandum NMFS-NWFSC-42. United States Department of Commerce, National Oceanic and Atmospheric Administration. Seattle, WA.

Milks, D., Varney, M, Schuck, M. and Sands, N. J. 2005. Lyons Ferry hatchery evaluation fall chinook salmon annual report: 2001 and 2002. Rept. to USFWS Lower Snake River Compensation Plan Office. April 2005. 100 p.

Morris, W.F. and D. F. Doak. 2002. Quantitative Conservation Biology: Theory and practice of population viability analysis. Sinauer Assoc. Inc. U.S.A. 480 p.

National Marine Fisheries Service. (1995). Proposed Recovery Plan for Snake River Salmon. U.S. Dept. of Commerce. March, 1995.

Wichmann, MC, K. Johst, M. Schwager, B. Blasius & F. Jeltsch. 2005 Extinction risk, coloured noise and the scaling of variance. Theor. Pop. Biol. 68: 29-40

Appendix B: Population Size and Complexity—Interior Columbia Chinook and Steelhead ESUs

Interior Columbia Technical Recovery Team
March 14, 2007

Background

The Interior Columbia Basin TRT has identified the basic population structure of these ESUs in a previous report. The tributary drainages used by populations within Interior Basin ESUs vary considerably in terms of size and complexity. Table B-1 summarizes the range in drainage area associated with Interior Basin ESU populations of Spring/Summer Chinook and steelhead. The intent of this analysis is to develop and apply an approach for characterizing the relative size and complexity of Interior Columbia Basin stream type chinook and steelhead populations based on available GIS data layers and empirically derived fish/habitat relationships. The results will be used by the Interior Columbia Technical Recovery Team to: 1) adapt viability curves (abundance/productivity criteria) to reflect population size, and; 2) contribute to the development of spatial structure/diversity criteria

Population Size Categories

Interior Columbia basin tributary habitats accessible to anadromous salmonids vary considerably in their ability to support spawning and rearing. We assigned populations identified within each of the ESA listed Interior Columbia ESUs to size categories based on an analysis of the amount of habitat that could support spawning and associated juvenile rearing. The following sections summarize the methods used to generate estimates of population size and to identify population size categories. The resulting population assignments are also summarized for each ESU.

Table B-1. Relative size (tributary drainage area) of extant populations within Interior Columbia Basin listed stream type Chinook and steelhead ESUs.

ESU	**Extant Populations (#)**	**Basin Drainage Area (km^2)**	
		Smallest	**Largest**
Snake River Spring/Summer Chinook	29	130	4,400
Upper Columbia Chinook	3	1,100	4,700
Snake River Steelhead	24	630	6,800
Mid-Columbia Steelhead	17	600	9,600
Upper Columbia Steelhead	3 (+1?)	1,100	4,700

Examples of populations occupying smaller drainages include Asotin Creek and Sulphur Creek (Snake River Steelhead and Spring/Summer Chinook ESUs); Rock Creek and Fifteen Mile Creek (Middle Columbia Steelhead ESU) and the Entiat River (Upper Columbia Steelhead and Spring Chinook ESUs). Populations using relatively large, complex tributaries include Upper John Day steelhead, Wenatchee and Methow River steelhead and spring chinook; and Lemhi River steelhead and spring/summer chinook. This natural variation in size and complexity suggests that even historically, populations likely varied in their relative robustness or resilience to perturbations.

Estimating Historical Population Size

We developed a method for assigning a relative weight to stream reaches based on physical characteristics (Appendix C). Using GIS layers, we mapped the physical characteristics for each 200 m reach within the tributary habitat associated with specific chinook and steelhead populations and assigned a weighted intrinsic potential using a simple model based on available measures of physical habitat characteristics. That model is driven by estimates of stream width, gradient, and valley width derived from a GIS-based analysis of the tributary habitat associated with each population. Each accessible 200-m reach within the tributary habitat associated with a specific population is assigned an intrinsic productivity rating based on the particular combination of physical habitat parameters listed above. Four categories were used: high, moderate, low, and not rated or zero potential. For application to yearling type chinook, sufficient information was available to add a negligible category. A weighted estimate of the total amount of rated habitat historically available to each population was constructed by summing the habitat by rating category, multiplying each sum by a relative weighting factor (1 = high, .5 = moderate, and .25 = low), and totaling the weighted sums. For this calculation, reaches rated as negligible were assigned a relative weight of zero.

Assigning Populations to Size Categories

Populations of stream type chinook and steelhead were tabulated (by species) in order of estimated total weighted stream kilometers of rearing habitat. Four general groupings of populations (Basic, Intermediate, Large and Very Large) were identified based upon relatively large increases in weighted spawning habitat between adjacent pairs of populations in the ordered list (Figure B-1).

We adapted the approach to accommodate the biological characteristics and available data for Snake River Fall Chinook and Snake River Sockeye populations, respectively.

Spring and Spring/Summer Chinook Size Categories

Basic Size Category: Chinook

A group of the smallest populations was identified based on a relatively large gap in relative size between the estimates for the Entiat and Chamberlain Creek populations. The median estimate of weighted historical spawning area for this category was 230,000 sq meters. Populations in this size category were relatively simple in terms of spatial structure (Table B-2).

Intermediate Size Category: Chinook

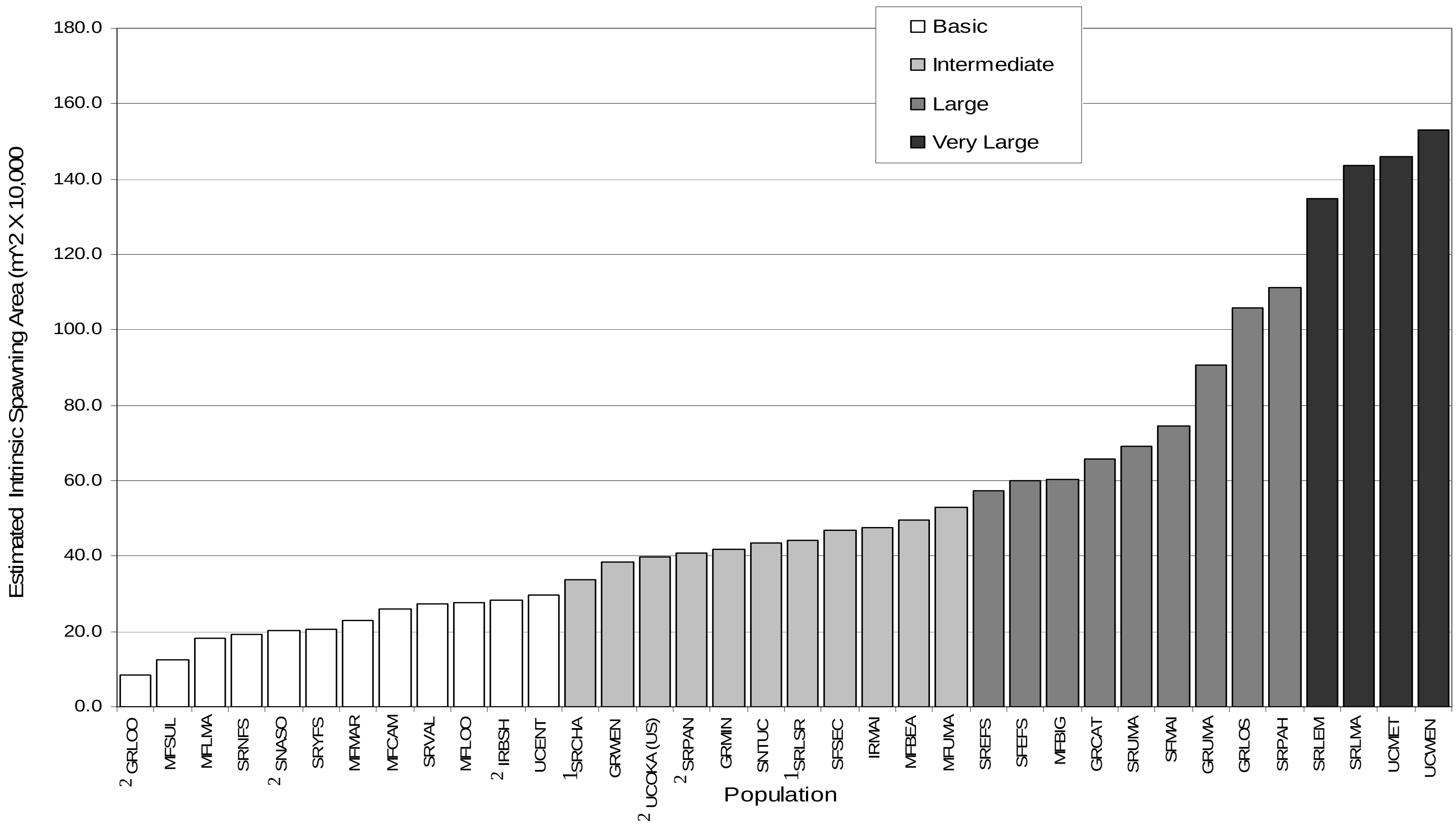
A grouping of 10 extant populations of intermediate size and complexity was defined by the breakpoints separating the groups of smaller and larger populations. The proportional range in population size within each of the three groupings was relatively consistent, with populations varying in size by roughly a factor of two (Table B-2).

Large Size Category: Chinook

Nine extant Spring/Summer Chinook populations were identified in a third grouping—the Large population size category. The median size (based on estimated historical potential) of populations in this category was roughly twice the median for the Intermediate size category. Populations in this category were also relatively more complex—the median number of Major Spawning Areas for populations in this category was three, compared to a median of one for the Intermediate size category (Table B-2).

Very Large Size Category: Chinook

The four largest extant populations were assigned to this size category. The principle difference between populations in this category and those in the Large category was overall size (weighted spawning area). The median size of populations in this category was twice that of the populations in the Large size category. The median number of MaSAs for this size category (four) was greater than that of the other categories (Table B-2).



[1]Abundance and productivity for these populations can be evaluated against the minimum abundance threshold for the next lowest size category level based on the amount of historical habitat in the core tributary area.

[2]Population is extirpated or functionally extirpated.

Figure B-1. Interior Columbia Basin Stream Type Chinook populations ordered by intrinsic potential (km of weighted spawning/rearing habitat). Bar shading distinguishes the different size categories (Basic, Intermediate, Large, Very Large).

Table B-2. Spring and Spring/Summer Chinook extant populations' (Upper Columbia Spring and Snake River Spring/Summer ESUs) summary statistics for population size categories. Estimates are based on the ICTRT historical intrinsic potential analysis.

Stream Type Chinook Populations		**Tributary Spawning Habitat—Population Size Categories**			
		Basic	**Intermediate**	**Large**	**Very Large**
Number of extant populations in the category		9	10	9	4
Spawning area (X 10,000 m2)	Median	23.0	43.9	69.2	144.8
	Range	(12.5-29.6)	(33.9-52.8)	(57.2-111.1)	(134.8-153.0)
Relative density at threshold abundance					
	Spwners/10,000m^2	21.7	17.1	14.4	13.8
	Ratio to Basic		0.79	0.67	0.64
Number of Major Spawning Areas per population	Median	1	1	3	4
	Range	(0-1)	(1-3)	(1-5)	(3-5)

Steelhead Size Categories

Steelhead tributary population areas were generally larger than the areas associated with Spring/Summer Chinook, reflecting the wider range of spawning conditions characteristic of steelhead. We identified four groups of steelhead populations based on 'breaks' in the cumulative size distribution across the forty seven populations incorporated into the analysis of historical potential. The four size groupings were generally reflected in our basic measure of within population spatial structure—the number of MSAs. (Figure B-2; Table B-3).

Basic Size Category: Steelhead

A group of the smallest populations was identified based on a relatively large gap in relative size between the estimates for the Joseph Creek and Touchet River (Walla Walla basin) populations. The median estimate of weighted historical spawning area across the 13 populations in this category was approximately 141,000 sq meters. Populations in this size category were relatively simple in terms of spatial structure (Table B-3). The median number of MaSAs per population in this category was one.

Intermediate Size Category: Steelhead

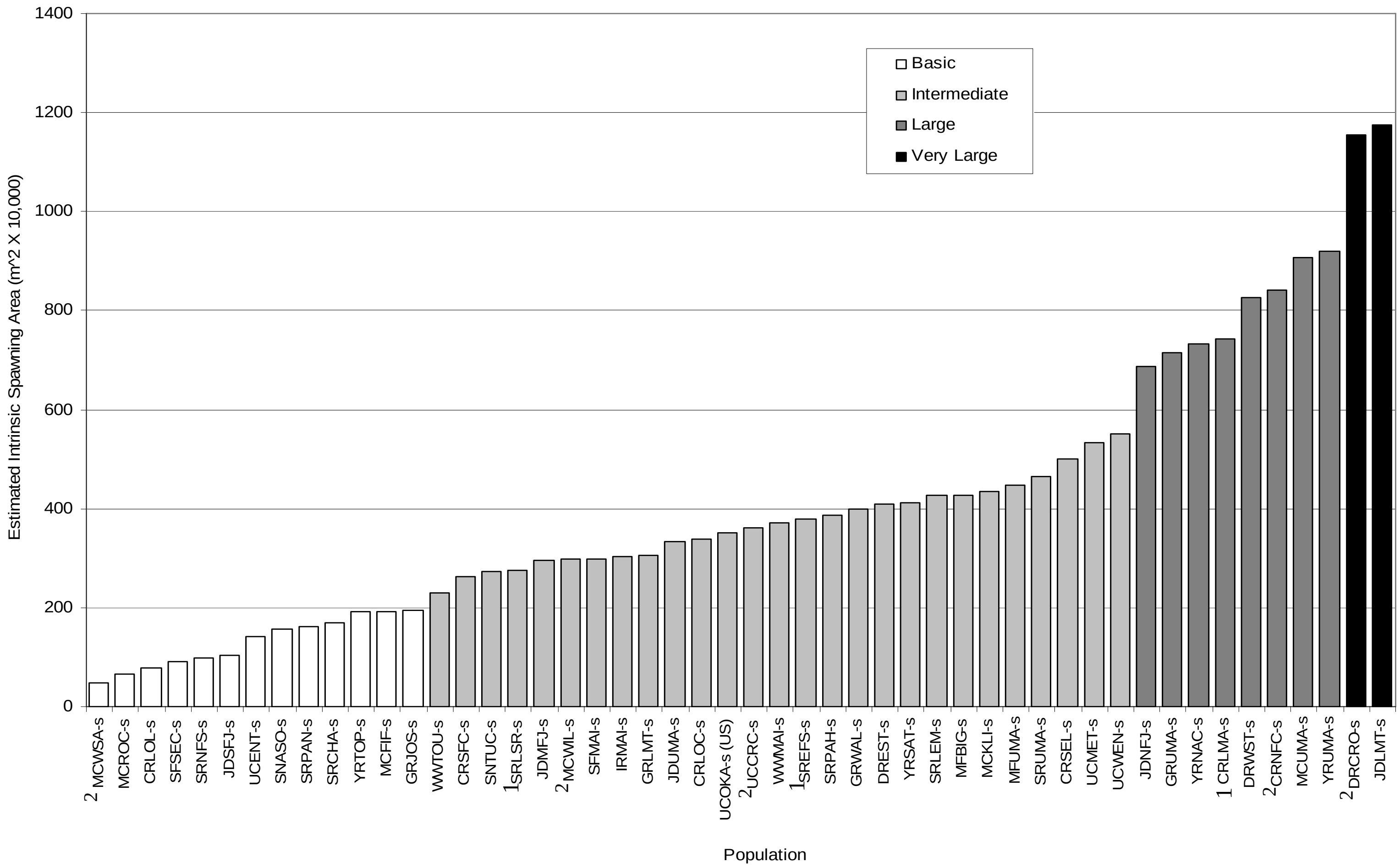
This category contained the largest number of extant populations (24). The Touchet River was the smallest population in this category. The Wenatchee River population defined the upper end of this category based on a relatively large increase to the next population in sequence (North Fork John Day River). The median population size for this category was 382,000 sq. meters, more than double the relative amount of habitat for the Basic category (Table B-3).

Large Size Category: Steelhead

Seven extant steelhead populations were identified in a third category, the Large population size category. The median size of populations in this category (743,000 square meters) was roughly twice the median for the Intermediate size category. The seven extant populations in this category are characterized by relatively high spatial complexity—the median number of MaSAs per population was eight (Table B-3).

Very Large Size Category: Steelhead

The largest extant population, the Lower Mainstem John Day River, was assigned to this category (the extirpated Deschutes Crooked River population also belongs to this category). The principle difference between populations in this category and those in the Large category was overall size (weighted spawning area). The spawning area of this category was approximately 1.5 times that of the populations in the Large category (Table B-3).



[1]Abundance and productivity for these populations can be evaluated against the minimum abundance threshold for the next lowest size category level based on the amount of historical habitat in the core tributary area.

[2]Population is extirpated or functionally extirpated.

Figure B-2. Interior Columbia Basin Steelhead populations ordered by intrinsic potential (km of weighted spawning/rearing habitat). Bar shading distinguishes the different size categories (Basic, Intermediate, Large, Very Large).

Table B-3. Steelhead extant population (Upper Columbia, Middle Columbia and Snake River ESUs) summary statistics for population size categories. Estimates are based on the ICTRT historical intrinsic potential analysis.

Steelhead Populations		Tributary Spawning Habitat—Population Size Categories			
		Basic	**Intermediate**	**Large**	**Very Large**
Number of extant populations in the category		12	25	7	1
Spawning area (X 10,000 m2)	Median	141.0	382.3	743.0	1175.4
	Range	(66.6–193.7)	(229.3–550.5)	(686.7–921.0)	(1175.4-1175.4)
Relative density at threshold abundance	Spwners/10,000m^2	3.5	2.6	2.0	1.9
	Ratio to Basic	---	0.74	0.57	0.54
Number of Major Spawning Areas per population	Median	1	4	8	13
	Range	(1-3)	(1-7)	(5-14)	(13-13)

Table B-4. Size category assignments for steelhead populations organized by ESU. Populations with substantial areas subject to possible temperature limitations are identified.

ESU	MPG	Basic	Intermediate	Large	Very Large
Upper Columbia Steelhead	***Upper Columbia Steelhead***	Entiat[2]	Wenatchee[2] Methow[2] Okanogan (US portion)[2]		
Middle Columbia Steelhead	***Cascade Eastern Slope Tributaries***	Fifteenmile[2] Rock[2] White Salmon[1]	Deschutes Eastside[2] Klickitat	Deschutes Westside	Crooked River[1]
	John Day River	JD South Fork	JD Upper Mainstem JD Middle Fork	JD North Fork	JD Lower Mainstem[2]
	Umatilla and Walla Walla		Walla Walla[2] Touchet[2] Willow[1]	Umatilla[2]	
	Yakima River Group	Toppenish[2]	Satus[2]	Upper Yakima Naches[2]	
Snake River Steelhead	***Lower Snake***	Asotin[2]	Tucannon[2]		
	Clearwater River	Lolo	Lochsa South Fork Selway	Lower Mainstem[2] North Fork Clearwater[1]	
	Grande Ronde	Joseph	Lower Grande Ronde Wallowa	Upper Grande Ronde	
	Salmon River	Chamberlain Panther Secesh North Fork	Lemhi Upper Salmon East Fork. Upper Salmon Mainstem Upper Middle Fork Lower Middle Fork Pahsimeroi Little Salmon[2] South Fork		
	Imnaha River		Imnaha		
	Hells Canyon	Hells Canyon Tributaries[1,2]			

[1]**Population is extirpated.** [2]**Potential for extensive temperature limitations (>10% of intrinsic spawning kms)**

Other Size Category Considerations for Stream Type Chinook and Steelhead Populations

Temperature limitations

The population size categories were based on physical measures of habitat—stream gradient and width were the determining factors for steelhead spawning potential. Other factors can substantially affect the relative productivity of a particular reach or watershed, including temperature conditions and aquatic productivity. We do not have a comprehensive data set representing historical (pre 1850) stream temperatures for Interior Columbia tributaries. We used regression models based on available stream temperature-elevation data to characterize reach specific temperature regimes. Those projections reflect the factors driving stream temperatures during the periods of observation and are not necessarily representative of historical conditions. However temperature mapping based on those relationships can be used to identify populations that are subject to relatively high stream temperatures during key rearing (and spawning periods).

Incorporating a summer temperature maximum constraint (weekly maximum less than 22 deg. C) substantially reduced the estimated amount of spawning habitat for many Mid-Columbia ESU and lower Snake River steelhead populations (Table B-4). In most cases the reductions in spawning area were associated with lower Mainstem small tributaries. The intrinsic spawning or rearing potential estimates for populations exhibiting relatively high potential temperature impacts should be validated using alternative information wherever possible.

Core Area Considerations

Many populations include mainstem and tributary habitat between core spawning reaches and adjoining downstream populations. In some cases these dispersed habitats contain a significant proportion of total intrinsic weighted area, but may have provided only limited connectivity between populations. The ICTRT summed weighted intrinsic potential for both the total population and for core spawning watersheds. If it was determined that; 1) limited connectivity existed; and 2) the core-only sum fell into a lower size category, then the minimum threshold was adjusted downward to reflect a more realistic biological scenario.

Four populations were affected by the core area considerations described above:

- Chamberlain Creek Spring/Summer Chinook population—minimum threshold abundance can be set at 500 (Basic) or 750 (Intermediate)

- Little Salmon River Spring/Summer Chinook population—minimum threshold abundance can be set at 500 (Basic) or 750 (Intermediate)

- Westside Deschutes River Steelhead population—minimum threshold abundance can be set at 1000 (Intermediate) or 1500 (Large)

- Little Salmon River Steelhead population—minimum threshold abundance can be set at 1000 (Intermediate) or 500 (Basic)

<u>Wide Mainstem Considerations</u>

Our estimates of historical population size are based on the intrinsic potential analysis described in Appendix C. The habitat ratings in that analysis were largely derived from empirical data reflecting the relationships between spawning abundance and physical habitat conditions in tributaries less than 15m to 20m in width. We extended the ratings to cover wider tributary mainstem type habitats based on relatively sparse empirical data. Wide mainstem type habitat is a significant component of the total intrinsic potential for some steelhead populations. The ability to support spawning in these sections may depend on additional environmental conditions not included in our intrinsic habitat model. The potential for historical spawning in these areas should be explicitly considered in population specific assessments. Populations with substantial mainstem habitat include the Okanogan in the Upper Columbia ESU; Satus Creek and the Lower John Day in the Mid-Columbia ESU; and the Lochsa, Little Salmon River and Lower Mainstem Clearwater River in the Snake River ESU. In addition, the mainstem Yakima River above the confluence of Satus Creek flows through an alluvial basin. Historically the combination of braided mainstem habitats and the constant input of relatively cold groundwater could have supported substantial production. Given the tributary origin of our criteria, this particular combination is not assigned a high rating. The possibility that this extensive reach may have supported substantial production should be taken into account.

Population Minimum Abundance Thresholds by Size Category

Because populations with fewer than 500 individuals are at higher risk for inbreeding depression and a variety of other genetic concerns (McClure et al. 2003 discusses this topic further), the ICTRT does not consider any population with fewer than 500 individuals to be viable, regardless of its intrinsic productivity. Therefore we set the threshold level (minimum acceptable long term average spawning abundance) for the smallest category of drainages at 500 spawners.

Incrementally higher spawning abundance thresholds were established for the remaining three population size categories (Table B-5). Increased thresholds for larger populations promote achieving the full range of abundance objectives including utilization of multiple spawning areas, avoiding problems associated with low population densities (e.g., Allee effects) and maintaining populations at levels where compensatory processes are functional. We set thresholds for the Very Large and Large size categories so that the expected average density at threshold abundance would be approximately ½ the density associated with 500 spawners for the median Basic population. Threshold levels for application to populations in the intermediate size category were set so as to achieve median spawner densities at approximately half the range between the median population size for Basic and Large population categories.

The approach of assigning incremental minimum thresholds based on four categories of population size represents a balance between two alternatives for setting a minimum abundance consistent with the objectives described above. One approach would be to set the minimum abundance threshold at 500, the number corresponding to preserving genetic characteristics assuming a randomly intermixed spawning population. Larger populations are generally more complex in terms of watershed structure. A minimum abundance threshold of 500 in a large population would translate to average densities of spawners much lower than 500 spawners in a Basic sized population. In addition, the basic assumption of a randomly intermixed population inherent in calculating the minimum estimate of 500 spawners would be questionable given the complexity of larger populations. Metapopulation effects associated with relatively low numbers of spawners in isolated sub watersheds of many populations would likely result in substantial increases in risk for larger populations at a minimum abundance level of 500.

An alternative approach that would emphasize equivalent seeding levels across tributary habitat would be to set the minimum threshold for smaller populations at 500 and set minimum abundance thresholds for the remaining populations at levels proportional to the relative amount of tributary spawning habitat. As an example, the Wenatchee population is approximately 8 times the historical tributary habitat relative to the median Basic sized population; the minimum abundance threshold for the Wenatchee would be set at 8 X 500, or 4,000. Under this approach, the resulting average spawning density within a population would be constant across populations with more historical habitat than the median sized Basic population. The minimum abundance levels set for larger populations would arguably correspond to substantially reduced risks relative to the corresponding levels for Basic or Intermediate populations.

Table B-5. Minimum abundance thresholds by species and historical population size (spawning area) for extant Interior Columbia Basin stream type chinook and steelhead populations. Median weighted area and corresponding spawners per kilometer are provided for populations in each size category.

Population Size Category	**Stream Type Chinook** (Upper Columbia Spring, Snake Spring/Summer ESUs)			**Steelhead** (Upper Columbia, Middle Columbia & Snake River ESUs)		
	Threshold	**Median Weighted Area (m X 10,000)**	**Spawners per KM (weighted)**	**Threshold**	**Median Weighted Area (m X 10,000)**	**Spawners per KM (weighted)**
Basic	**500**	23	21.7	**500**	141	3.6
Intermediate	**750**	44	17.1	**1,000**	371	2.7
Large	**1,000**	69	14.4	**1,500**	784	1.9
Very Large	**2,000**	145	13.8	**2,250**	1165	1.9

Fall Chinook and Sockeye Population Sizes

We established minimum abundance thresholds for stream type Chinook and steelhead populations based on our empirical intrinsic potential analyses and generalized minimum population recommendations from the literature. We do not have the same level of comparative habitat production potential information for fall Chinook (historically dominated by stream type production) or sockeye. We have established relative size categories for fall Chinook and sockeye populations consistent with our recommendations for yearling type Chinook and steelhead, incorporating recommendations from previous recovery planning efforts for those Snake River ESUs (NMFS, 1995).

Fall Chinook

Snake River fall chinook exhibit important life history differences relative to yearling Chinook and steelhead. Snake River fall Chinook spawned primarily in large mainstem reaches and the dominant juvenile life history pattern was for subyearling migration.

The ICTRT has designated three historical populations of Snake River fall Chinook, two of which occupied areas above the Hells Canyon dam complex, a total block to anadromous migration. The two extirpated populations represented the bulk of historical production within this ESU.

The intrinsic habitat potential analysis described in attachment B was developed based on empirical information for ocean type chinook and steelhead populations. The specific biological information used in analysis do not directly apply to the relationship between habitat conditions and spawning/rearing use by Snake River fall chinook. The ICTRT adapted the approach for identifying major and minor spawning areas as follows to reflect biological characteristics of Snake River fall chinook.

The current fall chinook run is predominately associated with Snake River mainstem habitat between the upper end of the Lower Granite Dam reservoir (near Asotin, Washington)and Hells Canyon Dam. That section of the Snake River mainstem is approximately 163 km in length and can be classified into three distinct reaches based on physical characteristics (Groves and Chandler, 1999). The uppermost reach, from Hells Canyon dam downstream to the mouth of the Salmon River, is characterized by a relatively narrow channel with short, deep pools interspersed with rapids. The middle reach, between the Salmon and Grand Ronde River confluences, widens considerably from a relatively narrow canyon section at its upper end and is characterized by lower gradients. Flows in this reach are augmented by the inflow from the Salmon River drainage. The lowest of the three mainstem reaches extends from the confluence with the Grand Ronde to the upper end of Lower Granite Pool. This reach is characterized by a wide channel with low shorelines, deep pools and relatively few rapids. Flow and turbidity are the most variable in this reach.

We evaluated recent redd distribution data in the context of the physical conditions described above. Redd distributions indicate a consistent gap (encompassing the middle reach as described above) in mainstem spawning between the confluences with the Salmon and Grand Ronde Rivers. Based on the distribution of physical habitat characteristics and the patterns in redd deposition, we defined two historical major spawning areas (MaSAs) in the mainstem Snake River upstream of the Lower Granite Reservoir (Figure B-3). One mainstem MaSA extends from the confluence of the Clearwater River upstream to the confluence of the Salmon River. The second mainstem MaSA extends from the confluence of the Salmon River upstream to the general vicinity of Hells Canyon Dam. We concluded that each of these mainstem reaches has the physical capacity to support a minimum of 500 spawners (extrapolated from habitat analyses in Connor et. al, 2001 and Groves & Chandler, 1999). Historically, there may have been an additional relatively contiguous reach capable of supporting spawning in the lower section of the Snake mainstem now inundated by the three lowermost Snake River dams.

The lower reaches of the five major Snake River tributaries entering the mainstem below Hells Canyon dam have been surveyed for fall chinook spawning in recent years. Significant numbers of redds have been located in three tributaries (the Clearwater, Tucannon and Grand Ronde River). Based on physical conditions and current redd densities, we conclude that all three of these lower tributary reaches should be considered as MaSAs in assessing the Snake River fall chinook population status. Although the core spawning area for this population was the mainstem, the alternative spawning locations in the lower mainstems of tributary rivers provide alternative sources of production when mainstem conditions are poor (e.g., low flows and/or high turbidity).

> *Based on these evaluations, the extant Snake River fall chinook population includes five MaSAs: the two mainstem reaches described above along with the lower reaches of the Clearwater, Grand Ronde and Tucannon Rivers. The lower reaches of the Imnaha and Salmon Rivers may have supported relatively low levels of fall chinook spawning and are considered part of the upper mainstem MaSA.*

We established a minimum abundance threshold for the extant fall chinook population consistent with the general abundance/productivity objectives summarized in the July 2003 ICTRT Viability Draft Report. We adapted the recommendations summarized in NMFS (1995) to assign a minimum long term average spawning abundance threshold for the extant population. We are recommending a minimum abundance threshold of 3,000 natural origin spawners for the extant Snake River fall chinook population. No fewer than 2,500 of those natural origin spawners should be distributed in mainstem Snake River habitat.

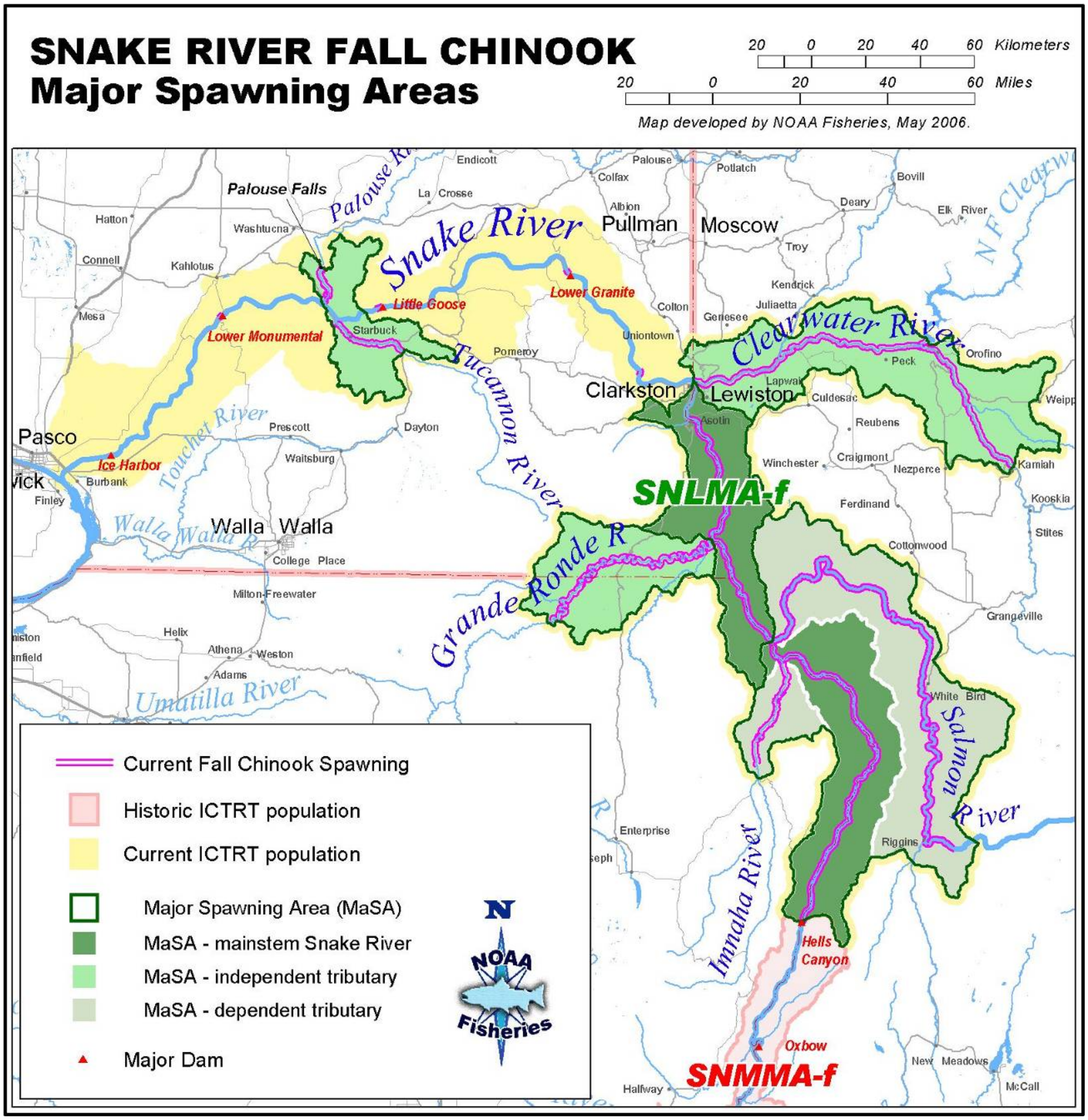


Figure B-3. Snake River Fall Chinook population. Current distribution of spawning areas.

Sockeye

Snake River sockeye have declined to extremely low levels and are currently associated with a single lake in the Stanley Lakes Basin. In previous TRT analyses (ICTRT 2003, McClure et al. 2005) we have concluded that at least three lakes in the Stanley Lakes Basin supported independent sockeye populations (Redfish Lake, Alturas Lake and Stanley Lake). Two other small lakes (Pettit Lake and Yellowbelly Lake) may have supported sockeye production, however currently available information is insufficient to support definitive conclusions regarding whether or not they supported additional sockeye populations.

Sockeye production is believed to be generally related to lake area, although other factors (e.g., temperature regime, relative aquatic productivity) strongly influence production levels (e.g., Burgner, 1991). Historically, sockeye production was supported in a number of lakes throughout the Columbia Basin (Gustafson et al. 1997, Waples et al., 1991). These lake systems varied considerably in size (Figure B-4). Sockeye supporting lakes in the Columbia basin can be classified into four categories based on estimated historical surface areas. The smallest size category (less than 250 hectares surface area) includes most of the Stanley Basin lakes along with Suttle Lake (Deschutes drainage). Alturas Lake and Redfish Lake fall into a second category along with Lake Wenatchee (Upper Columbia). A number of lakes outside of the Stanley Basin have current surface areas ranging from 1500 to approximately 2500 hectares. In addition to the lakes included in Figure B-4, there were several much larger lakes in Canada that have been substantially increased in area due to impoundments, including Lake Okanogan and the Arrow Lake complex. Each of these lake systems most likely exceeded 10,000 hectares in surface area. These systems constitute a fourth surface area category.

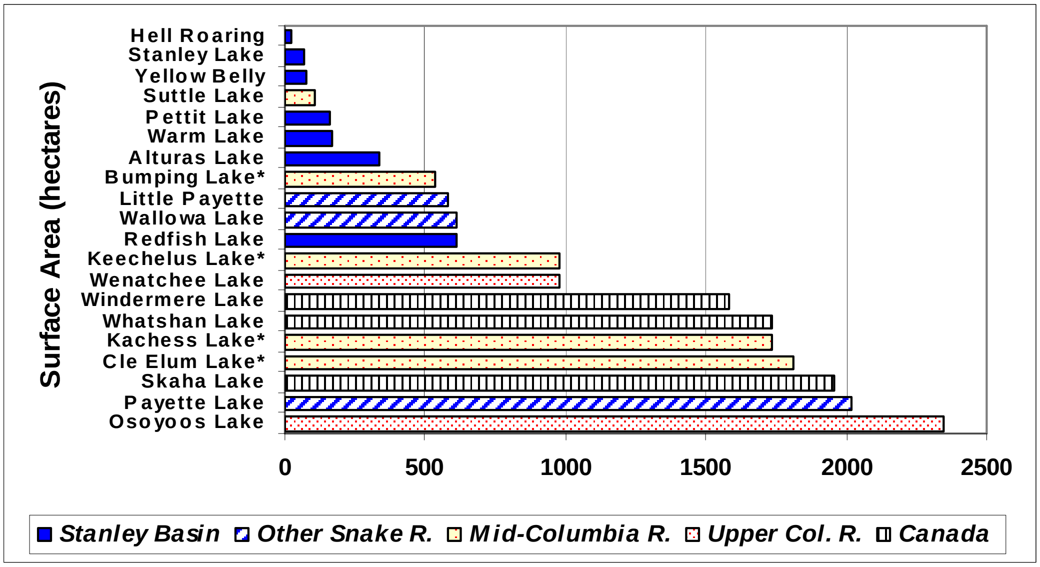


Figure B-4. Surface area (hectares) of lakes within the Columbia basin (not including major Canadian reservoirs impounded by dams). Solid bars: Snake River sockeye ESU ICTRT designated populations (Stanley Lakes basin in Idaho). Dashed fill bars: Possible additional Stanley Lakes basin historical sockeye populations. Dotted fill bars: Columbia basin lakes outside of the Stanley lakes basin currently (or historically) supporting sockeye production. Asterisk indicates current area expanded as a result of dam.

Defining Within Population Structure

Spatial structure varies greatly both within and among ESA defined chinook and steelhead populations. Both temporal and geographic variations exist within occupied systems, resulting in a wide array of spawning configurations. These structural differences have implications for a population's intrinsic viability, and by analyzing spatial composition, planners have an opportunity to evaluate how sustainable production can be achieved.

In our approach for describing spatial structure, we designated the basic building block for salmonid populations as a *branch.* In our definition, a branch component can be any reach organization containing suitable spawning habitat within a sub watershed. The quantity and interrelatedness of branches within a watershed contribute to a population's level of risk in regards to sustainable production.

Additionally, the organizational variation and quantity of branch habitat within targeted populations determine the distribution of Major (MaSA) and Minor (MiSA) Spawning Areas. A rule set (Figure C-2) was developed in order to clearly define and delineate MaSA and MiSA structure. As with branches, it is crucial to understand the geographic composition of spawning areas, and their associated implications, to manage for sustainable productivity.

Moving Window Methodology

Branch development

Using GIS techniques, we developed a methodology for defining and displaying branches. We applied a *moving window* design for evaluating habitat within steelhead and chinook ESA reaches. Our moving window spatial parameters were inherited from minimum branch size definitions, which are equivalent to the amount of habitat required to sustain 50 spawners (approximately 1.25 km for spring/summer chinook, and 3.0 km for steelhead). These stream distances, then, became the calculated lengths for our moving window spatial theme.

Using linear referencing techniques, we compiled tabular descriptions for the moving window features (Table B-6). Each window was addressed with a "from," "to," and feature code attribute. The addresses were offset by 200m increments, so that for each reach, the window began at 0m and stopped at 3000m (steelhead) or 1250 m (Chinook), and then continued upstream at 200m, ending at 3200m (steelhead) or 1450 m (Chinook). This pattern continued until the headwaters of the hydrologic feature were reached. The result was a set of overlapping segments representing a *moving window* spatial theme (Figure B-5).

Table B-6. Address table for linear referencing of “moving windows.”

FEATURE ID		BRANCHING PARAMETERS			
LLID	STREAM NAME	FROM CHINOOK(m)	TO CHINOOK(m)	FROM STEELHEAD(m)	TO STEELHEAD(m)
1190674487624	Pettijohn Creek	0	1250	0	3000
1190674487624	Pettijohn Creek	200	1450	200	3200
1190674487624	Pettijohn Creek	400	1650	400	3400
1190674487624	Pettijohn Creek	600	1850	600	3600
1190674487624	Pettijohn Creek	800	2050	800	3800

The second step was to identify each window’s intrinsic values and calculate an average rating. The mean intrinsic calculation was our fundamental metric for determining which widows qualified for *branch* status. Because our definition stated that branches could only contain “high” or “moderate” values (and hence, the most productive habitat), it was necessary to determine the average intrinsic rating and attribute it to individual windows. We achieved this by intersecting our moving window features with those from our intrinsic potential analysis, and then summarizing the mean rating for the segments underlying each window. From this analysis, we queried for where the mean intrinsic value was at least equal to “moderate” and saved it as a new spatial theme. In this way, our moving windows are represented as a spatially derived moving average of intrinsic habitat quality.

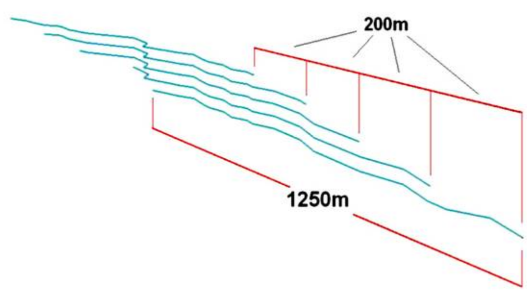


Figure B-5. Example of Spring Chinook “moving window” linear referencing

Designation of Major and Minor Spawning Area

Once our branched distribution was spatially defined, we delineated MaSA and MiSA subwatersheds. Major spawning areas were defined as a system of one or more branches that contain sufficient habitat to support 500 spawners. For Spring/Summer Chinook, this value was 100,000m^2, and for steelhead it equaled 250,000m^2. We generated area values by using hydrology tools within the GIS. Most commonly, these tools are utilized for calculating hydrographic features such as flow direction and accumulation, and watershed delineation.

In our evaluation, we employed flow accumulation functions (using the weighted area calculations from the intrinsic analysis) to calculate potential salmonid production. Starting from the highest elevation within a hydrologic basin, the aggregation continued downstream, accumulating branch habitat until the watershed outlet was reached. This technique produced a hydrologically accumulated grid which was weighted by the quantity of moderate and high intrinsic habitat within our previously defined branches. Using spatial analyst, we then subtracted the topographically derived (unweighted) flow accumulation from the intrinsically weighted accumulation grid. These results were then divided by 250,000 (steelhead) or 100,000 (Chinook). The values in the resulting grid illustrated where the minimum habitat criteria for MaSAs were met, so that each increasing whole number identified a new potential MaSA (dependent upon other criteria within the rule set). With both branches, and MaSA/MiSA minimums defined, the rule set was applied in order to define individual MaSA (or MiSA) subbasins.

Literature Cited

Burgner, R.L. 1991. The life history of sockeye salmon (Oncorhynchus nerka). In: C. Groot and L. Margolis (eds.), Life history of Pacific salmon. p. 3 -117. Univ. of British Columbia Press. Vancouver, B.C.

Connor, WP, A.P. Garcia, A.H. Connor, E.O. Garton, P.A. Groves and J.A. Chandler. 2001. Estimating the carrying capacity of the Snake River for fall chinook salmon redds. Northwest Science. 75(4): 363-371.

Groves, P.A. & J.A. Chandler. 1999. Spawning habitat used by fall chinook salmon in the Snake River. No. Amer. J. Fisheries Management. 19:912-922.

Gustafson, R.G., T.C. Wainwright, G.A. Winans, F.W. Waknitz, L.T. Parker and R. S. Waples. 1997. Status review of sockeye salmon from Washington and Oregon. U.S. Dept. of Commer., NOAA Tech. Memo. NMFS-NWFSC-33.

ICTRT 2003. Independent Populations of chinook, steelhead and sockeye for listed Interior Columbia Basin ESUs. Draft. Interior Columbia Basin Technical Recovery Team Report. July, 2003

ICTRT 2005 – pop id update memo.

National Marine Fisheries Service. (1995). Proposed Recovery Plan for Snake River Salmon. U.S. Dept. of Commerce. March, 1995.

Waples, R.S. 1991. Pacific salmon, *Onchorhynchus* spp., and the definition of species under the Endangered Species Act. Mar. Fish. Rev. 53(3):11-22.

Appendix C: Interior Columbia Basin Stream Type Chinook Salmon and Steelhead Populations: Habitat Intrinsic Potential Analysis

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March 16, 2006

Introduction

Interior Columbia River Basin (ICB) salmon and steelhead have evolved to take advantage of a wide diversity of habitats. Climatic, geological, topographic, and landcover patterns have produced a robust evolutionary trajectory in streams flowing through vastly disparate terrestrial environments. This opportunity for uniquely adapted populations has created a challenge for identifying, both qualitatively and quantitatively, intrinsic habitats within large watersheds such as the ICB. Though salmon and steelhead occupy streams flowing through a wide spectrum of upland environments, their freshwater habitat preferences are limited to a comparatively narrow set of hydrological and streambed conditions (Reiser and Bjornn, 1979). However, it is the interaction between apposite flow path structure and adjacent terrestrial geomorphologies that determines intrinsic suitability. Ultimately, site specific stream reach characteristics and salmonid habitat preferences are influenced negatively and positively by both adjacent and out of view landscapes.

The analysis described below is intended to provide a simple and objective overview of the distribution of historical production potential across the tributary habitats used by Interior Columbia basin yearling type Chinook and steelhead populations. The initial iterations of our approach were patterned after an analysis of Puget Sound Chinook habitat potential developed by the Puget Sound Technical Recovery Team. That approach relied on empirically derived relationships between salmon spawner densities and channel characteristics (Montgomery et al., 1999). In the Puget Sound Chinook application, production potential was expressed in terms of spawners per unit reach length and related to a set of physical reach level measures: stream width, stream gradient, valley width and vegetative cover. In combination these factors were related to the relative amount of pool habitat, an important determinant of relative spawning and juvenile density. Similar sets of reach level habitat measures have been used to map relative production potential for coho and steelhead in Oregon coastal watersheds (Nickelson, et al., 1992, Burnett, 2001) and for steelhead in the Willamette River drainage (Steel, 2004).

Methods

We developed a reach level intrinsic potential (IP) analysis for application to stream type Chinook and steelhead spawning reaches assess habitat quality within currently and historically occupied portions of the ICB. This approach has enabled us to formulate a baseline perspective from which we can assess contemporary changes to productivity. Utilizing established relationships between habitat type, stream structure, landscape processes, and spawning use, we built a locally adapted Geographic Information System (GIS) based model incorporating regional spatial data, fisheries surveys, and professional knowledge. The GIS was used for the development, presentation, management and modeling of spatially referenced data. Modeled geomorphological characteristics were assigned to unique categories comprised of gradient, width, and valley confinement, from which additional stream and landform modifiers were incorporated to adjust intrinsic potential. We then evaluated these classes against known distributional densities in order to test modeled habitat quality. Results from these comparisons were used to weight and summarize reach areas for the entire stream network within the ICB based on relative Chinook salmon and steelhead habitat preferences.

We used the following process to develop the historical intrinsic potential analysis for Interior Columbia basin tributary habitats:

1. Fish density vs. habitat characteristics: Reviewed literature and available data sets relating simple measures of habitat characteristics to production potential for salmon and steelhead.

2. GIS data acquisition: Acquired and developed GIS data describing key habitat measures related to salmon and steelhead production potential for ICB ESU populations as determined in step 1.

3. Determining boundaries: Identified and applied criteria for defining the upper and lower boundaries to Chinook salmon and steelhead production within ICB watersheds using natural barrier locations and other habitat factors.

4. Initial classification: Classified stream reaches based on habitat characteristics (stream width, gradient, valley confinement) into categories representing varying levels of relative productivity. These habitat classes where then used to attribute spawning reaches, with respect to modeled salmon and steelhead production potentials, as high, moderate, low, negligible or none.

5. Preliminary validation and updating: Compared results from step 4 against specific measures of relative abundance of spawning adults and provided output to regional fisheries biologists for review. Additional habitat factors (reflected in GIS layers) were incorporated into the IP analysis to improve the correspondence of modeled distributions with empirical data and field observations.

6. Finalizing and applying reach level ratings: Finalized relative spawning potential rating categories as a function of physical habitat characteristics, and generated weighted totals by population and associated sub areas.

Fish Density Data Analysis

Our preliminary efforts focused on identifying published data and reports that related simple measures of habitat characteristics to stream type Chinook salmon and steelhead production. We found that direct measures of life stage specific productivity within particular reach characteristics are rarely available at fine scales or distributed across multiple watersheds. In fact, there is no single dataset with a consistent measure of relative abundance across the full range of environmental conditions found within ICB streams. As a result, we based our investigation on a set of discrete regional data sets. In general, we utilized spawning surveys, habitat studies, and stream transect juvenile sampling data to describe relative densities of stream type Chinook and steelhead in geospatially specific stream reaches.

Juvenile Abundance Transects

Initially, analyses relating densities of juveniles measured at a consistent life stage to habitat characteristics were used to assign relative intrinsic potential ratings and identify important structural elements within stream reaches. Studies generally show that for both yearling and stream type Chinook, juvenile densities are typically highest in relatively low gradient, unconfined stream reaches with well defined pool structure (e.g., Hillman& Miller, 2002, Petrosky & Holubetz, 1988), while steeper gradient relatively confined tributary reaches typically support the highest relative densities of juvenile steelhead (e.g., Slaney et al., 1980, Petrosky & Holubetz, 1988, Burnett, 2001). Steelhead have also been reported to use braided mainstem reaches for spawning and rearing, given appropriate flow, temperature and substrate conditions (e.g., ODFW, 1972).

Idaho Parr Data. Using juvenile transect survey data collected by the Idaho Department of Fish and Game (IDFG), we completed additional analyses comparing juvenile abundance to stream habitat. In the early to mid 1980's, IDFG biologists compiled a baseline data set for evaluating the effectiveness of habitat improvement projects. The data set included both measures of parr densities (Chinook and steelhead/rainbow trout) and habitat measures. The IDFG studies (as concluded (as discussed above) that Chinook parr densities were the highest in low gradient stream sections in relatively wide valleys and that steelhead/rainbow juvenile densities were the highest in steeper gradient, more confined reaches (e.g., Petrosky & Holubetz, 1988). The original analyses focused on data collected in years with relatively high parental escapements to minimize the confounding effect of relatively low seeding (Petrosky and Holubetz, 1988). We used data from naturally seeded areas from that parsed data set for the current analyses. For stream type Chinook (figure 1) and steelhead (figure 2), parr densities were plotted against gradient and stream width within two valley width categories corresponding to B channel and C channel designations (Rosgen, 1985) used in the original study. We found that wider stream reaches known to be used for spawning and rearing by steelhead were not well represented in the Idaho baseline study. A second data set, compiled by the Washington Department of Game for larger rivers in western Washington and Puget Sound, was also analyzed to provide some insight into production relationships in larger systems.

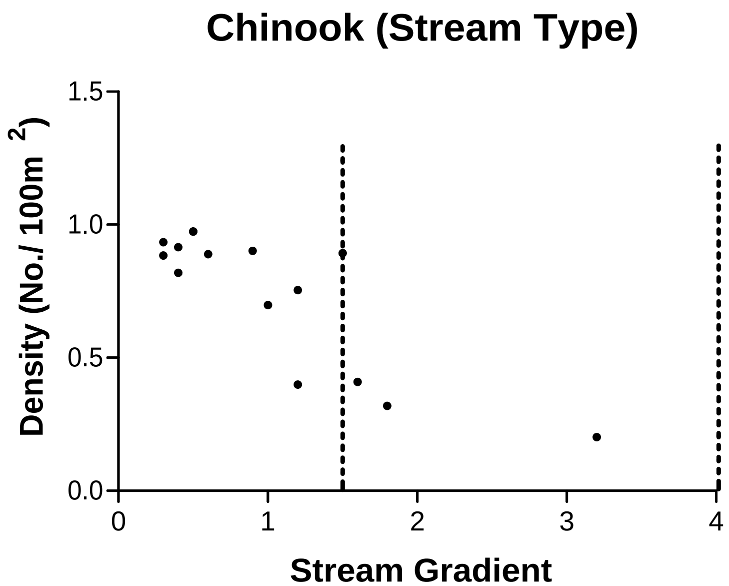

Figure 1. Idaho Spring/Summer Chinook. Juvenile densities vs. stream gradient for naturally seeded baseline monitoring areas in the Salmon and Clearwater River systems. Parsed data set—low seeding years not included (Petrosky and Holubetz, 1988). Dotted lines indicate assigned category boundaries.

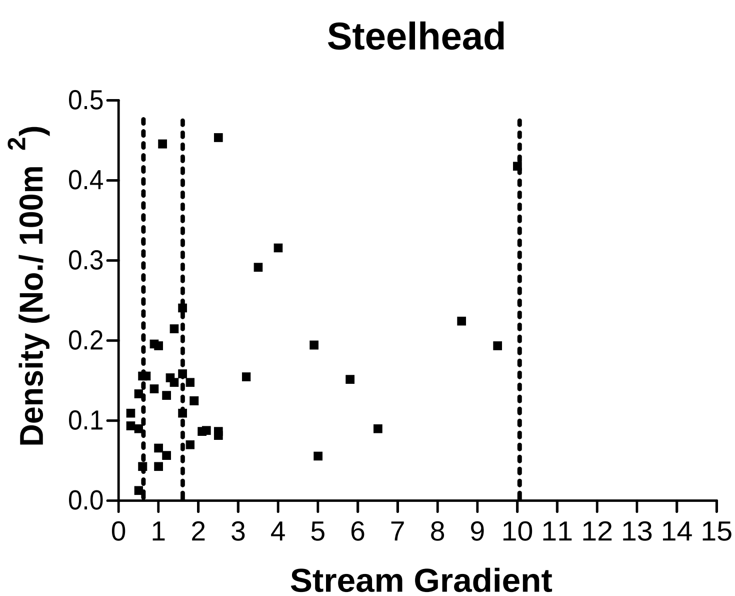

Figure 2. Idaho Steelhead. Juvenile densities vs. stream gradient for naturally seeded baseline monitoring areas in the Salmon and Clearwater River systems. Parsed data set- low seeding years not included (Petrosky and Holubetz, 1988). Dotted lines indicate assigned category boundaries.

The results from these investigations became the foundation for our habitat modeling scheme and helped identify the structural elements that would be required for additional analyses. Specifically, it became quite apparent that accurate measures of stream width, gradient, and valley confinement would be crucial for assessing intrinsic potential within the GIS. Developing models and acquiring data that describe these variables at a reasonable scale became our next task.

GIS Data Acquisition and Modeling

The National Hydrography Dataset (NHD) 1:100,000-scale networked reach model was used as the base stream layer for our intrinsic potential analysis. The NHD's layer contains all hydrographic features, including naturally flowing reaches and anthropogenic constructs such as irrigation canals, ditches, and laterals. Using only natural flow paths from the networked data, we built a linearly referenced stream layer comprised of contiguous 200-meter stream reaches. Segments were *addressed* using a "from", "to", and "id" field by dividing each unique stream into a continuous set of 200-meter tabular entries (stream length / 200 = number of events per stream), from which linear referencing processes were used to geocode address attributes within the hydrography network. This segment length was chosen to facilitate our classification of salmonid barriers, as a 200-meter reach with a 20% gradient has been found to be impassable for upstream migrants (Cramer, 2001; WDNR, 2002). These 200-meter hydrosections have become the basic unit of measurement for all ICTRT intrinsic potential summaries and analyses.

<u>Stream Gradient</u>

Stream gradient has been found to be an important habitat qualifier for salmonid spawning preference, and is determined by the change in vertical distance over reach length. As a flow path characteristic, gradient functions both as an indicator of upstream limit on migration (Cramer, 2001; WDNR, 2002) and as a predictor of habitat quality within accessible reaches (Cramer, 2001; Lunetta *et al*., 1997). Within the GIS, we used linear referencing techniques and zonal statistics to generate elevation values for all 200-meter stream segments. The minimum (downstream-most point) and maximum (upstream-most point) stream elevations were calculated using the USGS's National Elevation Dataset (NED) 10-meter horizontal resolution digital elevation models (DEMs).

Although spatial agreement is relatively high between the NHD's 100k hydrography and the NED, we had to augment standard neighborhood analysis techniques recognizing that even small misalignments can introduce large errors into the gradient calculations. We developed a procedure using Euclidean geometry to assign elevations for each segment in order to resolve the relatively small geographic differences between the DEM flow paths and our NHD derived 200-meter reach segments. Within each stream length, 10 equally spaced positions were linearly referenced to the reach and were given a unique code. We then calculated a contiguous zone for each point and computed a zonal statistical summary comparing the Euclidean output to the DEM. From these data, the minimum value determined for each zone was assumed to be the elevation of the DEM flow path, and therefore assignable to the vector stream layer for computational accuracy. An additional summary was generated for each unique 200-meter stream segment in order to obtain the minimum and maximum value from the previous calculation that used intervening points. Using the measures from this output as the upstream and downstream elevations, we attributed all linear features with their computed gradient.

Channel Bankfull and Wetted Width

Stream widths are an important metric for determining the amount of available habitat and the upstream extent of migrants. In our analysis, we have utilized both bankfull and wetted widths as a means of recognizing spawning time differences between stream type Chinook and steelhead. Because steelhead spawn near the peak of the hydrograph, and conversely, stream type Chinook salmon spawn near its lowest point, it was more accurate to assign different stream dimensions for both species. Therefore, we have applied bankfull width to steelhead and wetted width to stream type Chinook salmon, and all measurements relating to specie specific habitat totals include these adjustments in the calculations.

Stream width is predominantly a function of stream discharge, which can be estimated from a combination of drainage area and precipitation (Leopold *et al.*, 1964; Sumioka *et al*. 1998). Therefore, utilizing discharge as a proxy for stream width, we estimated stream dimensions from watershed size and mean annual precipitation. We used measured widths from field based stream measurements within the Columbia River basin to develop equations for estimating bankfull and wetted width (ODFW, 1999; WDOE, 2004). Upstream drainage area and accumulated average annual precipitation for each width measurement were derived from 60-meter DEMs (resampled from the 10-meter NED) and a 4-km grid of mean annual precipitation (1971-2000) (NCDC, 2004).

We conducted an analysis using linear regression between measured stream width and the accumulated precipitation and basin size metrics. For bankfull width, we applied the appropriate channel measurement within the field data; for wetted width, only measurements taken during August and September were included to accurately represent stream type Chinook salmon spawning times. Both analyses yielded statistically significant relationships between the basin size, precipitation, and stream width values and the resulting regression model was applied to the 200-meter reach data.

Valley Confinement

We estimated mean valley width for each reach by projecting 20 transects across the DEM-defined valley floor in each 200-m segment, and then calculating the mean valley width of the segment. The horizontal extent of the transect (valley width) was determined using flood height calculations from previous studies (Hall, 2007). As with our gradient calculations, we accounted for spatial discrepancies between the NHD 100k streams and the DEM flow path by calculating floodplain width based on the DEM flow path, and then assigning the calculated floodplain width to the 200-meter stream segments for subsequent data analyses.

Specifically, the valley width was calculated by creating a Euclidean based layer whose value was inherited from and spatially centered to the flow path elevation for each transect. Additionally, the flood height value was added to this grid layer, and the resulting calculation was subtracted from the NED. The results in this output grid showed the extent of the floodplain (based on the assigned flood height) where the values were less than or equal to zero. These valley areas were then summarized for all 20 transects independently, from which a mean value was generated and attributed to each 200-meter segment.

Determining Upstream and Downstream Extents

Upstream limits on the potential use of tributary habitat for spawning and rearing by salmon and steelhead were defined in terms of physical barriers, stream gradient, width, and water temperature. Reaches above documented natural obstructions and DEM calculated gradient barriers were excluded as production areas. Stream reaches with gradients above 5% were also excluded as spawning/rearing areas for yearling Chinook salmon populations based on expert opinion and on a review of index reach data sets for ICB streams. Minimum stream widths capable of supporting spawning were estimated based on available width measurements for index reaches with documented redd counts and mapped distributions. Additionally, a water temperature model was used to mark the downstream extent of spring Chinook salmon in Upper Columbia and Lower Snake River populations.

Natural Barriers

Barrier identification was our first data development scheme describing habitat quality, and employed both GIS calculated gradient barriers (representing the 20% limit described previously), and documented features such as falls, cascades, and reaches disconnected by sub-surface flows. We have utilized multiple digital, hardcopy, and field personnel sources to determine where natural obstructions mark the upstream extent of salmon and steelhead habitat. When possible, GIS datasets describing barriers were identified and incorporated into the base layer. In many cases archived report material and expert opinions had to be transferred to digital media and spatially referenced using recorded locations (such as river distance or an identifiable landmark). We have converted all sources of information into a GIS point feature theme and have preserved narratives and source information.

Within our IP analysis, natural barrier identification has been an ongoing process. Some features previously identified as complete barriers have been removed due to inconsistent information (such as salmon or steelhead observations above these locations) and others have been labeled as variably accessible due to significant year to year changes in stream flow, and hence passability. Local review of ICTRT data has provided many new additional barriers, which have been used to update stream accessibility metrics. In all cases, we have identified the 200-meter segments adjacent to complete migration blockages and have attributed all corresponding upstream features as inaccessible habitat.

Stream Width

Stream channel size generally decreases as you move upstream. At some point, stream dimensions constrict to such a point that habitat becomes unusable for salmon and steelhead. For spring Chinook, we used two data sets in order to determine stream size limitations; results from recent USFWS redd mapping efforts in the Middle Fork Salmon River, and Grande Ronde redd count index reaches. For steelhead, we utilized John Day redd count index reaches, O. *mykiss* presence/absence data from ODFW, IDFG parr count transects from the Salmon and Clearwater basins, and suitability maps developed by IDFG (Thurow, 1988). Channel widths calculated for the 200-meter segments used in the IP analysis were spatially joined to each dataset, and mean values were summarized for each unit. In both the spring Chinook and steelhead analyses, we used the 95th percentile low value for bankfull and wetted width to delineate our upstream extent. Use of smaller tributaries for juvenile rearing has been documented (e.g., Nez Perce tribal comment letter), and spawning in smaller tributaries may occur in particular situations. Further discussion of our stream width metrics will follow in the next section.

Water Temperature

The lower reaches of many interior basin tributaries are subject to summer temperatures that are well above levels injurious to salmon and steelhead. Persistent high temperature levels can have a significant impact on the ability of a given reach to sustain both juvenile rearing and adult spawning. Although current thermal regimes within ICB drainages are significantly influenced by human activities, it is likely that some lower reach habitat has always been temperature limited. Unfortunately, there are no temporally or spatially broad datasets describing historical temperature profiles, so any model using contemporary data reflects current habitat degradations. This is important to note, because any modeling exercise which uses current data will have output shaped by modern externalities.

A Streamnet (1999) temperature dataset was used for modeling water temperatures as they relate to environmental characteristics. We adopted the temperature criteria used by Chapman & Chandler (2001) which determined that a weekly mean average temperature (WMAT) exceeding 22 degree C could potentially limit or exclude salmon and steelhead production. Using NCDC mean July temperatures (1971-2000), percent forest cover (calculated from USGS NLCD), and elevation (USGS DEM), we developed a reach specific model that predicts the likelihood of exceeding a WMAT of 22 degree C. In the Streamnet dataset we chose data points that were the least likely to be anthropogenically altered. These included locations directly above or below dams, within irrigation infrastructures, or adjacent to urbanized areas. The final analysis revealed significant relationships between a WMAT of 22 degree C and air temperature, percent forest cover, and elevation. These variables were used to develop a simple screen that either included or excluded 200-meter segments within the 22 degree C zone. This delineation was then used to define the lower extent of spring Chinook salmon spawning potential in Upper Columbia River and Lower Snake River Populations. It should be noted that the initial set of variables used in this analysis do not reflect the effects of groundwater on ameliorating temperatures in mainstem reaches with broad, alluvial flood plains such as those found in the Lower Yakima River.

Reach Level Habitat Potential Ratings

Four different habitat measures were used to define our criteria for estimating reach specific production potential for stream type Chinook and steelhead within ICB habitats. The characteristics selected were; (1) stream width (modeled as bankfull and wetted width), (2) stream gradient (change in elevation over reach length), (3) valley width (relative width of valley compared to bankfull width) and (4) riparian vegetation (as a percent of landcover). We previously discussed how these variables were calculated using a GIS, and will now describe the methods employed for categorizing data.

Stream Width.

We established three stream width categories after considering the range of widths associated with the empirical density data for Interior Columbia streams, the relative distribution of channel widths in areas identified as supporting steelhead spawning in the basin and the categories employed in the Puget Sound analysis. The three categories were 3.6 m(wetted) or 3.8 m(bankfull) to 25 m, 25 - 50 m and >50 m. The rationale for our upstream extent (minimum stream width) was described earlier, and agrees with other observations. For example, streams less than 3 m in bankfull width were at the lower margins sampled in the Idaho baseline study. Also, presence/absence data provided by the Nez Perce Tribal staff indicates that few streams less than 3 m support production for steelhead. WDFW has recommended using a 2 m wetted width as the lower limit for steelhead in western Washington streams. Although most transects within the Idaho parr data were between 3.8 m and 25 m bankfull width, the WDG study included mainstems up to 50 m wide, and this value defines the upper limit of our moderately sized width class. Very little abundance data existed for the largest mainstem rivers (>50 m).

Based on previous analyses, we set lower limits relative to spawning/rearing potential of 3.6 m (wetted width) for Chinook and 3.8 m (bankfull width) for steelhead. Spring Chinook spawn in the late summer and early fall, and summer wetted width is an appropriate measure of stream size relative to this time period. Steelhead spawn in the late spring on the end of the spring freshet, and bankfull width is a more appropriate measure of stream size relative to this period.

Valley Confinement

The Idaho baseline study classified streams as B or C type channels using criteria defined by Rosgen (1985). Using the valley confinement estimates calculated earlier, we defined 200-meter reaches within our IP analysis as C type if valley width exceeded 20 times bankfull width. Values less than 20 times bankfull width were either attributed as confined or unconfined (defined below).

Confined streams with moderate to high gradients are unlikely to exhibit the stream structures necessary to support salmon and steelhead spawning. We incorporated a measure of confinement (as a function of valley to bankfull width) into our IP criteria, and assigned categories to all 200-meter segments. Streams that have a valley to bankfull width ratio less than 4 are defined as confined, and have virtually no opportunity for lateral channel migration and floodplain development (Beechie *et al*., 2006, Hall *et al*., 2007). This means that confined channels lack instream processes which promote the development of suitable spawning substrates. If valley width was less than 4 times bankfull width, a stream segment was attributed as confined and the intrinsic production potential was downgraded by one level.

Gradient

A set of gradient categories was developed based upon the Puget Sound TRT Chinook matrix (e.g., Table 2 in WRIA 18 Draft Summary Report - Puget Sound Chinook Recovery Analysis Team) and the categories used in the Idaho and Washington Game Department studies. For Chinook, most of the observed parr density/stream gradient data pairs fell within the 3 to 25 m stream width category. In general, densities were relatively high at gradients below 1.0 to 1.5 %. Although observations were relatively sparse, densities were low at gradients exceeding 1.5 to 2.0 percent. The frequency of samples exhibiting low pool cover (less than 50%) increased rapidly as gradients exceeded 1.5%.

Steelhead exhibited the reverse pattern with relatively low densities at gradients below 0.5, increasing as gradients rise to approximately 4%. Steelhead parr densities remained relatively high as gradients increased above 4%. We assigned the highest potential rating to gradients between 4% and 7% (an upper limit consistent with expert opinion cited in the draft Lower Columbia/Willamette TRT Viability report). Stream reaches in the 3.8-25 m bankfull width category that had gradients between 7 and 15% were designated with low potential. No spawning potential was assumed if gradients exceeded 15%. Steelhead parr densities at gradients exceeding 1.0 remained at relatively high levels in the widest streams in the sampled areas, but transects located in streams greater than 20 m bankfull width were not well represented.

We used adult steelhead spawning surveys to supplement the parr data analyses in determining relative ratings for streams exceeding 25 m bankfull width. Klickitat River index redd counts (YKFP 2002) and radio tracking results for Yakima Basin steelhead (Hockersmith et al., 1995) were geo-referenced and used to describe width and gradient classes in spawning locations within larger streams. We modified our ratings for the 25-50 meter wide category using the relative ratios generated from these analyses.

Riparian Vegetation

An additional modifier was originally incorporated into the framework based on forest cover as a source of large woody debris (LWD). Using the USGS (2000) National Land Cover Dataset (NLCD), we calculated the percent of forest within buffered 200-meter stream segments, and classified reaches with greater than 90% forest cover as mesic forest. In Puget Sound stream systems (PSTRT 200?), pool structure is affected by the availability of large woody debris (LWD), which can mitigate for the limitations of moderate gradient reaches. Initially, we included the assumption that LWD sources within adjacent riparian areas (classified as mesic forest) would result in increased pool structure in moderate gradient reaches (and would therefore increase suitability). However, analysis of the USFWS Middle Fork adult redd data set did not support increased production potential (redd densities) in forest versus non-forested reaches in moderate gradient or confined reaches. As a result, we dropped this rating category from our analysis.

Initial Rating Assignments

Classes assigned to stream gradient, width (bankfull and wetted), and valley confinement were grouped into habitat categories and given a rating of “high”, “moderate”, “low”, or “none.” These relative ratings were determined from observed life stage specific abundance values within specific habitat classes and applied to the 200-meter stream segments within our IP dataset. Maps from this exercise were distributed to regional biologists for review.

Review and Modification Including Additional Habitat Screens

The results from our habitat suitability classification were analyzed using two methods: solicited reviews from field biologists and comparisons with current spawning survey summaries. Firstly, maps were developed for individual watersheds and distributed to local agencies for review and comment. Feedback from this process then became the basis for developing sediment and stream velocity habitat screens as they relate to intrinsic quality. Secondly, statistical comparisons were made between IP habitat classes and productivity as measured by redd counts. The spring/summer Chinook survey from the Middle Fork Salmon River (USFWS) was used for our IP analysis of stream type Chinook, and WDFW steelhead surveys in the Upper Columbia (2004-06) were used to compare with O. mykiss IP values. Both datasets were important because they included redd surveys of entire streams, making non-occupied reaches significant and comparable to IP modeled categories. Based on these comparisons, some class specific adjustments were made to IP ratings, most notably for adding confinement as a significant feature in steelhead ratings, modification of gradient and width classes, and removal of the mesic forest modifier.

Habitat Screens-Sedimentation

The ability of a particular reach to support salmonid spawning can be significantly affected by sediment conditions within that reach (e.g., Bjornn and Reiser, 1991). Relatively low gradient stream reaches meandering through wide valleys can be deposition areas for fine sediments, especially if the surrounding soil types are highly erosive and fine grained. We used available GIS layers summarizing soil characteristics to assign relative indices of erosion potential and particle size to each tributary reach. The indices were calculated as an average across the HUC-6 corresponding to each particular stream reach.

Stream sedimentation is often a critical factor limiting the spatial distribution of salmonid spawning. In riverine systems, certain environmental traits promote the accumulation of stream sediments that can obscure suitable substrates. Specifically, the deposition of fine particles within streams is effected by factors such as soil type and hydrological conditions. In our analysis, these attributes were employed in order to determine where sedimentation might influence salmon and steelhead production. Most crucial to our investigation were the identification of highly erodible soils and low gradient streams which maximize particle detachment and limit transport.

Two primary data sources were utilized in our effort to locate probable sedimentation: the USDA-NRCS STATSGO soil survey, and reach level gradients obtained from USGS DEMs. The STATSGO dataset contains a measure of potential erodibility, or K factor, which is a predictive measure (0.0 – 1.0) of particle detachment resulting from rainfall. Soil texture and permeability are the key factors in determining the K factor, with clays having the lowest value (least erodible) and silts having the highest (most erodible). The USDA-NRCS considers soils with a K factor greater than 0.40 to be the most highly erodible and prone to runoff. Soils in this category are predominately composed of silts and silty loams. It should be noted that K factor is a measurement for bare soil conditions, and our analysis is for intrinsic habitats. However, natural disturbances would likely aid in the process of sedimentation more readily in soil units with the greatest erosion potential.

In addition to soil erodibility, we utilized stream gradients as a measure of depositional potential. Gradients were calculated for all 200-meter reaches within our study area using the minimum and maximum elevation per reach as obtained from the USGS DEMs. Low gradient streams result in lower flows and reduced stream power, which in turn promotes depositional rather than transport processes.

In order to determine stream reaches most at risk for sedimentation, we developed a habitat screening mechanism based on K factor and gradient. We first selected low gradient streams (<= 0.5%) and then intersected these results with soil units having a K factor greater than 0.4. Also, we identified sub watersheds having at least 50% of their area within highly erodible soils (K > 0.4). Low gradient reaches within these watersheds and those intersecting highly erodible soil units were attributed with high sediment potential. Additionally, the accumulated mean K factor was calculated for upstream reaches above all 200-meter segments, and where the accumulated mean was greater than or equal to 0.4 we applied the sediment screen. In reaches that were previously classified with moderate or high IP ratings, values within the sediment screen dropped to low.

Stream Velocity

For steelhead, an additional screen was developed in order to address highly rated IP areas identified as low potential by regional biologists. These reaches were primarily at the upper ends of drainages or emanated from relatively arid headwater areas. Generally, it appeared that persistent low flow conditions would preclude steelhead occupation. Using the NHD Plus database, we spatially joined mean annual stream velocity attributes to the 200-segments within the IP analysis. We then compared existing measure of productivity at specific locations (John Day steelhead index reaches, IDFG suitability maps, and Upper Columbia redd counts) to NHD calculated mean annual velocities and determined upper and lower limits. As with the sediment screen, all moderate and high potential rated reaches were changed to low if they were located outside the acceptable value range.

John Day Gravel Assessment-- stream confinement and gradient

Additional reviews from local biologists identified highly rated IP steelhead habitat within confined reaches and higher gradients that unlikely could support suitable substrate development. Stream gravel assessments within the Joseph Creek subwatershed were used to evaluate the significance of gradient and confinement to the distribution of suitable spawning substrates. The original dataset was developed by ODFW and was based upon stream surveys conducted in 1965 and 1966.

Spawning gravel summaries were classified by ODFW using "good" and "marginal" qualifiers, but the total of both categories were used for our analyses. We summarized mean bankfull width, confinement (valley width / bankfull width), and gradient for all 200 meter reach segments within the surveyed streams and joined it to the stream gravel dataset. The confinement parameter was expressed as the percent of stream confined (confinement was defined for reaches where valley width was less than or equal to 4 times bankfull width). To facilitate the standardization of gravel quantity among streams, the gravel area was divided by the bankfull stream area to compute the amount of gravel per unit stream area. These values were then multiplied by 10,000 to convert the values to integers.

We utilized an ANOVA to determine if there were differences between the amount of available spawning gravels within different gradient and confinement groups. Percent of stream confined was classified into two categories (<10% confined [uc], >10% confined [c]), and gradient was classified into 3 groups (0 – 1.5%, 1.5 – 4.0%, and > 4.0%). From the ANOVA, the streams with a greater percentage of confinement and higher gradients were shown to contain fewer spawning gravels as a percentage of stream area. These results were applied to our IP assessment by introducing confinement parameters to the steelhead habitat criteria.

Middle Fork Salmon and Upper Columbia Redd Surveys

The Middle Fork Salmon survey included GPS located redds within all accessible streams (1995-2003 return years, R. Thurow USFS pers. comm.). In the Upper Columbia (Okanogan, Methow, and Wenatchee subbasins), GPS data was collected (2004-2006) for redds observed in specific streams (C. Baldwin, WDFW pers. comm.) By identifying the nearest IP stream reach for each redd, we successfully quantified the total number observed per 200-meter segment in the intrinsic potential dataset. These results enabled us to evaluate our classification of IP habitat using observed redd densities by spatially joining predicted values to field measurements. Categories were summed by total Chinook or steelhead redds located within each habitat class, and an ANOVA was used to compare the total redd counts to unique categories. The results showed general agreement between our IP analysis (predicted quality) and redd density (observed productivity), but some differences were noted. These results were used to adjust model parameters to reflect spawning patterns observed for stream type Chinook in the Middle

Fork Salmon River and steelhead in the Upper Columbia, and formulated our final rating scheme.

Using the results from our ANOVA analyses, the greatest mean redd count for a habitat category was assigned a “high” intrinsic spawning potential. This group represented the most preferred habitat by observed Chinook and steelhead spawners in the dataset. Any grouping whose mean redd count was at least fifty percent of this highest value was also attributed with a “high” intrinsic potential. Continuing, those categories receiving between 25% and 50% of the highest value were given a “moderate” rating, between 12.5% and 25% a “low” rating, and less than 12.5% a “negligible” rating. The “negligible” rating was only applied to the stream type Chinook IP classification. These values were then used to weight potential habitat (for both area and length) so that a “high” rated reach was multiplied by 1.0, “moderate” by 0.5, “low” by 0.25, and “negligible” by 0.0. Functionally, the “negligible” category had the same effect on total habitat as inaccessible areas or those failing to meet our minimum width criteria (which were assigned a “none” rating). Neither the “none” or “negligible” classification contributed habitat, in terms of weighted length or area, to the total intrinsic spawning potential per population.

Species Specific Ratings

The final rating assignments are provided in Tables C-1 and C-2 for yearling type Chinook salmon and steelhead reaches, respectively.

Yearling Chinook

Table C-1. Relative potential for Interior Columbia basin Spring and Spring/Summer Chinook salmon spawning and initial rearing as a function of stream reach physical characteristics. BF: Bankfull stream width; Gradient: percent change over 200 m reach; and relative ronfinement: valley width expressed as ratio to BF stream width.

Stream Width/ Gradient Categories		**Valley Width Ratio (Ratio of valley width to bankfull stream width)**		
Bankfull Width (BF)	**Gradient**	**Confined** (<= 4 X BF width)	**Moderate** (4 to 20 X BF width)	**Wide** > 20 X BF width
BF < 3.7 m	≥ 0	None	None	None
BF 3.7 to 25 m	0 - 0.5	***Medium***	***High***	***High***
	0.5 - 1.5	***Low***	***Medium***	***High***
	1.5 - 4.0	***Low***	***Low***	***Medium***
	4.0 - 7.0	Negligible	***Low***	***Low***
	> 7.0	None	None	None
BF 25 m to 50 m	0 - 0.5	None	***Medium***	***Medium***
	0.5 - 10.0	None	None	None
	≥ 10	None	None	None
BF > 50 m	≥ 0	None	None	None

Steelhead

Table C-2. Relative potential for Interior Columbia basin steelhead spawning and initial rearing as a function of stream reach physical characteristics. BF: Bankfull stream width; Gradient: percent change over 200 m reach; and relative confinement: valley width expressed as ration to BF stream width.

Stream Width/ Gradient Categories		**Valley Width Ratio (Ratio of valley width to bankfull stream width)**		
Bankfull Width (BF)	**Gradient**	**Confined (<= 4 X BF width)**	**Moderate (4 to 20 X BF width)**	**Wide > 20 X BF width**
BF < 3.8 m	≥ 0	None	None	None
BF 3.8 to 25 m	0 - 0.5	None	***Medium***	***Medium***
	0.5 - 4.0	***Low***	***High***	***High***
	4.0 - 7.0	None	***Low***	***Low***
	> 7.0	None	None	None
BF 25 m to 50 m	0 - 4.0	***Low***	***Medium***	***Medium***
	> 4.0	None	None	None
BF > 50 m	≥ 0	None	***Low***	***Low***

Population Totals: Historical Potential Spawning Habitat

An estimate of potential spawning habitat area is a particularly relevant measure for use in expressing the size of specific populations relative to abundance and productivity criteria. A strong tendency for returning spawners to home back to natal spawning areas is a general characteristic of Chinook and steelhead. The predominant life history patterns for both of these species involve a year or more freshwater rearing, generally in the natal tributary. Returns to particular spawning reaches are therefore largely dependent upon the production from the previous generation of spawning in that same reach. As a result, the availability of suitable quantities of high quality rearing habitat also affects production and therefore average abundance associated with a particular spawning area.

Once final habitat adjustments were completed for the IP analysis, we weighted stream metrics using our new screening elements. In some cases, new criteria changed the rating by one or two categories, and in others the screen factor completely eliminated habitat potential (Table C-3). We used these updated results to generate population specific estimates of total spawning potential. We expressed the total amount of historical spawning habitat for each population as an equivalent amount of good spawning habitat. We weighted the amount of habitat (length and area) in each 200 meter reach within a population by a simple proportion corresponding to the assigned reach rating – high, medium, or low (we included a fourth category – negligible, for yearling type Chinook populations). Units of habitat rated with high production potential for a species were given a weight of 1. Units of medium production potential were given a relative rating of 0.5 and habitat units classified as low production potential were assigned a relative rating of 0.25. For Chinook populations, some reaches were rated as negligible. For the purposes of this analysis those reaches were assigned a weight of 0. A relative index of productivity for aggregate areas was calculated by summing the weighted total amounts of habitat within each category within the appropriate geographic units. The ratios of 1 to .5 to .25 for high, medium and low intrinsic potential categories reflect the patterns observed in the WDG steelhead parr density study (Gibbons et al., 1985, table 6) and are generally consistent with relative densities reported for spring Chinook late fall parr in the Idaho studies.

<u>Tributaries Supporting Two Chinook ESUs</u>

The intrinsic potential analysis described above is based on general physical requirements for Chinook spawning and early rearing. Some population areas in the Interior Basin support more than one Chinook ESU. We adjusted the total area assigned to the listed spring Chinook population in accordance with the following observations.

Upper Columbia Spring Chinook

Each of the extant populations of upper Columbia spring Chinook is associated with a population of summer Chinook. With the possible exception of the Entiat, summer Chinook runs are believed to have been endemic to each system. Upper Columbia River summer Chinook salmon are classified in a separate ESU. There are significant differences in life history patterns between the two ESUs - summer Chinook return to the Columbia River primarily in July and August, spawn approximately 1 month later than spring Chinook, and leave their natal tributary for the mainstem during the summer of their first year of life. Summer Chinook spawn later and lower down in the mainstems of the major Upper Columbia tributaries. Gradient and substrate characteristics of stream habitat within the stream sections used for spawning are similar for both runs. There is some overlap in each system between the lower end of the spring run spawning and the upper end of summer Chinook spawning.

Summer Chinook salmon utilize the Wenatchee River mainstem up through Tumwater Canyon for spawning. Spring Chinook salmon spawning is generally confined to the major tributaries to the Wenatchee and the mainstem reach downstream of Lake Wenatchee to Tumwater Canyon.

In the Methow basin, summer Chinook spawning is confined to the mainstem Methow River below the Chewuch River confluence (Anon., 1998). Chapman et al. (1994) states that summer/fall Chinook utilize the lower 50 miles of the Methow River mainstem. In the Okanogan, summer Chinook salmon currently spawn between Zosel Dam and the town of Mallott and from Enloe Dam to Driscoll Island.

Spring Chinook spawning in the Entiat drainage occurs above river mile 16 of the mainstem and in the lower five miles of a major tributary, the Mad River. Summer Chinook spawning extends downstream from approximately river mile 20 to the mouth.

Snake River Spring/Summer Chinook

There is limited potential for overlap in spawning/rearing areas among ESUs of Chinook in the Snake Basin.

Tucannon River: Currently, fall Chinook use the lower 10 km of the Tucannon mainstem for spawning (redd survey data summarized in Milk et al, 2005). Spring Chinook spawning currently occurs in the mainstem from the mouth of Sheep Cr. (river mile 52) downstream to King Grade (RM 21) - draft Lower Snake Recovery Plan p 82). The Tucannon system has been heavily impacted by human activities, resulting in increased stream temperatures and high sedimentation rates. Projections of historical temperatures indicate almost all of the mainstem Tucannon would have had average July temperatures below 22 deg. C.

Table C-3. Population total historical intrinsic potential spawning habitat. Units are 10,000 m^2 (equivalent to 1 km of 10 wide stream of reach habitat rated in High category). Core area habitat is the portion of the total within the major tributary drainage for the corresponding population.

Steelhead				Chinook			
ESU	Population	Total	Core	ESU	Population	Total	Core
Upper Columbia Steelhead	UCENT-s	141	136	Upper Columbia Spring Chinook	UCENT	30	30
	UCMET-s	533	526		UCMET	146	146
	UCWEN-s	550	488		UCWEN	153	153
	UCOKA-s (US)	352	336		UCOKA (US)	40	41
	UCCRC-s	360	---	Snake River Spring/Summer Chinook	SNASO	20	20
Middle Columbia Steelhead	MCWSA-s	48	46		SNTUC	44	44
	MCKLI-s	436	435		GRWEN	38	38
	MCFIF-s	191	164		GRLOS	106	106
	DREST-s	408	408		GRLOO	8	8
	DRWST-s	825	457		GRMIN	42	42
	MCROC-s	67	67		GRCAT	66	34
	MCWIL-s	298	255		GRUMA	91	91
	DRCRO-s	1156	---		IRMAI	48	48
	JDLMT-s	1175	1170		IRBSH	28	28
	JDNFJ-s	687	687		SRLSR	44	28
	JDMFJ-s	296	296		SFMAI	75	55
	JDSFJ-s	103	103		SFSEC	47	47
	JDUMA-s	335	335		SFEFS	60	60
	MCUMA-s	907	783		SRCHA	34	21
	WWMAI-s	371	360		MFBIG	60	60
	WWTOU-s	229	229		MFLMA	18	8
	YRTOP-s	191	157		MFCAM	26	26
	YRSAT-s	411	180		MFLOO	27	27
	YRNAC-s	734	535		MFUMA	53	53
	YRUMA-s	921	921		MFSUL	12	12
Snake River Steelhead	SNTUC-s	272	188		MFBEA	50	50
	SNASO-s	157	94		MFMAR	23	23
	CRLMA-s	743	743		SRPAN	41	40
	CRNFC-s	841	---		SRNFS	19	17
	CRLOL-s	78	78		SRLEM	135	133
	CRLOC-s	340	340		SRLMA	144	144
	CRSEL-s	500	500		SRPAH	111	111
	CRSFC-s	262	262		SREFS	57	57
	GRLMT-s	306	306		SRYFS	21	21
	GRJOS-s	194	194		SRVAL	27	27
	GRWAL-s	399	399		SRUMA	69	69
	GRUMA-s	714	714				
	IRMAI-s	304	304				
	SRLSR-s	276	85				
	SRCHA-s	169	60				
	SFSEC-s	92	92				
	SFMAI-s	299	299				
	SRPAN-s	163	125				
	MFBIG-s	428	428				
	MFUMA-s	448	448				
	SRNFS-s	98	62				
	SRLEM-s	426	368				
	SRPAH-s	385	257				
	SREFS-s	379	165				
	SRUMA-s	464	464				

Literature Cited

Burnett, K.M. 2001. Relationships among juvenile anadromous salmonids, their freshwater habitat, and landscape characteristics over multiple years and spatial scales in the Elk River, Oregon. Ph.D. Dissertation. Oregon State University, Corvallis, Oregon.

Gibbons, R.G., P.K.J. Hahn and T. H. Johnson. 1985. Methodology for determining MSH steelhead spawning escapement requirements. Washington State Game Dept. Report. #85-11. 43 p.

Grande Ronde Subbasin Spawning Gravel Abundance Survey 1965-66 [transcribed to computer file]. 2006. LaGrande, OR: Oregon Department of Fish and Wildlife.

Hockersmith, E., J. Vella, L. Stuehrenberg, R. Iwamoto, G. Swan. 1995. Yakima River radio telemetry study: steelhead, 1989-93. Dept. of Energy. Bonneville Power Admin. Fish and Wildlife Div. Project 89-089. 95p.

John Day River Steelhead Index Reach Redd Counts [computer file]. 1959-2003. Salem, OR: Oregon Department of Fish and Wildlife.

Montgomery, D.R., E.M. Beamer, G.R. Pess and T.P. Quinn. 1999. Channel type and salmonid spawning distribution and abundance. Can. J. Aquat. Sci. 56:377-387.

National Hydrography Dataset (NHDPlus) [computer file]. 2005. U.S. Environmental Protection Agency (USEPA) and the U.S. Geological Survey (USGS). Available WEB: http://www.horizon-systems.com/nhdplus/drainage-area.htm [December 6, 2006].

Petrosky, C. E. and T. B. Holubetz. 1988. Idaho habitat evaluation for offsite mitigation record. Annual report. 1987. Project 83-7. Dept. of Energy. Bonneville Power Admin. Fish and Wildlife Div.

Rosgen, D.L. 1985. A stream classification system. North American Riparian Conference. Tucson, Arizona. April 16-18 1985.

Steel, E. A., B. E. Feist, D. Jenson, G. R. Pess, M. B. Sheer, J. Brauner, R. E. Bilby. 2004. Landscape models to understand steelhead (Oncorhynchus mykiss) distribution and help prioritize barrier removals in the Willamette Basin, OR, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences, 61:999-1011.

Thurow, Russ. 1985. Middle Fork Salmon River fisheries investigations. Idaho Department of Fish and Game. Project F-73-R-6. 96p.

Wenatchee and Methow River Steelhead GPS Redd Surveys [computer file]. 2004-06. Wenatchee, WA: Washington State Department of Fish and Wildlife.

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MEMORANDUM

Date: January 8, 2007
From: Interior Columbia Technical Recovery Team
To: NMFS NW Regional Office, co-managers and other interested parties
Subject: Role of large extirpated areas in recovery

Summary and Conclusions

In this memo, the Interior Columbia TRT evaluates the role of extirpated Major Population Groups (MPGs) and populations in the functioning of listed ESUs in the Interior Columbia, as indicated in our viability criteria document (ICTRT, 2005). In our evaluation, we consider the potential contribution of the extirpated MPGs to ESU-level abundance, productivity, spatial structure and diversity in the context of the current and historical distribution of the ESU.

Restoring populations within currently extirpated MPGs to viability has the potential to increase the overall sustainability of several ESUs. However, predicting a quantitative benefit in risk reduction associated with re-establishment of populations in these areas is challenging and includes a high degree of uncertainty. Therefore, given the logistic challenges associated with re-introducing fish to many of these areas and the uncertainty of the contribution of re-established populations to ESU viability, we recommend a staged, adaptive approach to recovery planning and implementation. Such an approach gives highest priority initially to implementing actions within currently occupied areas and thus to improving the status of extant populations and MPGs. This approach emphasizes preserving existing genetic and phenotypic diversity. It does not suggest that historically occupied areas are not important to the ultimate long-term persistence of these ESUs, but rather that preserving extant populations should take temporal priority over reintroductions in situations where resources are limited. In this approach, recovery actions in currently occupied areas should be implemented concurrently with two supporting activities:

- A robust monitoring program, allowing evaluation of the likelihood of long-term persistence of the ESU when recovery goals in currently occupied areas are achieved.
- Scoping and planning for re-introductions into currently extirpated areas that would allow re-introductions to occur in a timely fashion when additional evaluations indicate that long-term persistence is dependent upon such re-introductions, or where they would be of most important to the viability of MPGs and ESUs

We concluded that the role of extirpated MPGs and populations varied from ESU to ESU as follows:

- Both Snake River fall chinook and Snake River sockeye are currently restricted to a single extant population. The probability of long-term persistence of both of these ESUs will be greatly enhanced with additional populations. In fact, these ESUs cannot meet the minimum ESU biological viability criteria established by the TRT without multiple viable populations.
- We have also concluded that viable populations within extirpated MPGs of the Upper Columbia spring chinook and steelhead ESUs would substantially increase the probability of long-term persistence of those ESUs.
- For the Snake River spring/summer chinook ESU, viable populations within the Clearwater ESUs would lower the overall risk to that ESU by improving the connectivity among extant MPGs and increasing the range of habitat types occupied by this ESU. However, due to the large number of populations and the spatial structure of the extant ESU, the relative contribution of these MPGs is somewhat lower than in cases where the extant ESU is more restricted.
- Viable populations within the extirpated areas of the Snake River steelhead ESU would lower overall risk, but likely not appreciably, again due to the large number of populations that are extant, and this ESU's current widespread spatial distribution.
- No MPGs are extirpated within the Mid-Columbia steelhead ESU. Extirpated populations and subpopulations within MPGs should be considered within the context of MPG and population viability.

An adaptive approach to recovery planning for extirpated areas

We are recommending that a step-wise, adaptive approach to these extirpated MPGs be taken due to uncertainties associated with reintroduction efforts.

The first consideration is uncertainty in quantifying ESU-level probability of persistence or risk of extinction or quasi-extinction. For example, simple metapopulation modeling efforts (e.g. Ruckelshaus et al. 2004) suggest that areas with fewer populations are at inherently greater risk than areas with more populations. However, quantifying the precise change in overall demographic risk is impossible, given uncertainty in a variety of factors including likely future environmental conditions, rates and impacts of potential catastrophic events, level of homing fidelity and likely historical distributions. Quantitative predictions are even less supportable when considering the biological benefits or costs of changes in components of ESU-level spatial structure and diversity. In most cases the diversity of those extirpated populations has been lost. The ability of introduced populations to restore some of that diversity is also highly uncertain. For these reasons, we describe the likely relative change in risk or likelihood of persistence that would result from the restoration of currently extirpated MPGs.

Re-introductions also are likely to have both initial or short-term effects and long-term benefits. In the short-term, they are unlikely to contribute substantially to abundance or productivity of the ESU. In addition, diversity benefits, particularly local adaptation, will require at least several generations to be realized. Similarly, the risk of outbreeding depression or introducing "domesticated" genes to neighboring populations is relatively high at the early stages of an introduction effort. At the low abundance and productivity that is likely in the initial stages, spatial structure benefits will also be minimal. However, in the long-term, as naturally-produced and locally-adapted populations become established, they can contribute to overall ESU abundance, productivity, and diversity. Finally, those populations will mitigate the risk of catastrophic loss, can provide connectivity between currently occupied populations and contribute to other natural interactions between populations.

Therefore, we recommend that initial, primary emphasis be placed on recovery of extant MPGs. In the case of ESUs with only one extant MPG, recovery actions should target the modified MPG risk levels defined for single-MPG ESUs in the July (2005) IC-TRT viability document. However, the potential that re-introductions will be necessary should not be neglected, particularly in those areas with the most potential for increased occupancy to improve ESU-level status. Concurrently with the implementation of recovery actions in currently occupied areas, a robust monitoring program should be implemented. This should be coupled with an ongoing evaluation or assessment of the likelihood of long-term persistence of the ESU as its status improves to determine whether re-introductions may be critical for long-term persistence. In addition, appropriate scoping or planning activities for re-introductions should occur, in the event that currently accessible habitat does not appear to be sufficient to assure the long-term persistence of the ESU. Appropriate scoping and planning activities include identifying suitable source broodstock for re-introduction, evaluating conditions in potentially accessible areas, improving those conditions if necessary, and other related activities that will improve the likelihood of a successful introduction.

Considering extirpated MPGs in ESU-level risk.

As with populations, ESU-level risk or probability of persistence is affected by abundance, productivity, spatial structure and diversity (McElhany et al. 2000). However, also as with populations, ESUs likely varied, even historically, in their inherent status. For example, the Upper Columbia spring chinook and steelhead ESUs appear presently and historically to contain fewer populations and MPGs than are currently occupied within the Mid-Columbia or Snake River steelhead ESUs. This simpler structure suggests that these smaller ESUs might have been at greater present or historical risk than some of the larger ESUs might be with the loss of one to several MPGs. The benefits of re-populating extirpated areas are thus dependent on this historical context.

We address three factors, described in our July (2005) document, that contribute to overall ESU viability for each extirpated MPG:

1. Demographic contribution of the MPG and its component populations to the ESU. This factor deals with the contribution of the MPG to the abundance and productivity of the ESU.
2. Spatial role of the MPG in the ESU. This factors deals with the contribution of the MPG to spatial processes, such as mitigating the risk of extinction due to localized catastrophes and ensuring normative demographic and genetic connectivity.
3. Contribution to overall ESU diversity. This factor deals with the likely degree of difference or variation likely to have been expressed by fish in the extirpated MPG.

We also considered the context of the extant and extirpated MPGs within each ESU, including:

- Total number of extant and extirpated populations
- Total number of extant and extirpated MPGs
- Total area available to the ESU historically and currently

In no case do data exist that allow us to evaluate the true contribution of currently extirpated areas to the ESU abundance, productivity, spatial structure or diversity. Thus, we used our analysis of likely intrinsic potential to evaluate several surrogate metrics as indicators. Specifically, we examined the number and proportion of stream kilometers (weighted by quality) that are currently accessible and that are no longer accessible as an indicator of contribution to ESU-level abundance and productivity. To assess the likely role of extirpated MPGs in ESU-level spatial structure, we estimated the distance from each MPG to its nearest neighbor under current and historical (i.e., all MPGs occupied) conditions. This process allowed us to evaluate quantitatively the likely role each MPG played in ESU-level connectivity. We evaluated the distribution of MPGs across the landscape (i.e., ensuring that some MPGs were relatively distant) qualitatively. We also qualitatively evaluated the risk of loss due to catastrophe. In particular, we anticipated that the presence of low-risk populations in multiple MPGs will reduce the risk of loss due to a single, local or sub-basin scale catastrophe, because we defined MPGs on the basis of geographic proximity and topographic and ecological similarity (and genetic similarity in currently occupied areas). Finally, to evaluate potential contributions to ESU diversity, we evaluated the distribution of high and moderate-quality stream kilometers across EPA ecoregions, using ecoregion as a proxy for potential phenotypic differences.

A. Snake River spring/summer chinook

Of the seven extirpated MPGs potentially belonging to this ESU, restoration of Dry Clearwater MPG would have the greatest impact on ESU viability, given the current number and distribution of occupied MPGs . Other extirpated MPGs would clearly contribute to ESU persistence (Table A-1) but the extant MPGs would likely be sufficient to ensure long-term persistence of the ESU if viability of those MPGs is achieved, due to the number, diversity and distribution of populations and MPGs that are currently occupied.

Chinook in the Clearwater River were extirpated by the construction and operation of Lewiston Dam in 1918. Stream-type chinook currently in the Clearwater basin are derived from Rapid River and other hatchery stocks. The current populations found in the Clearwater may provide some ecological functions within the ESU – particularly connectivity between the Lower Snake and Grande Ronde/Imnaha or Salmon River MPGs. Though not currently part of the Snake River spring/summer chinook ESU, these non-local fish offer a unique opportunity to evaluate both the efficacy of alternative re-introduction strategies and the rate and quality of local adaptation processes.

We also evaluated the possibility that there might have been one or more ESUs above the current Hells Canyon dam complex historically (see Population Identification update memo; further discussion to be provided in final Population Identification document). Unfortunately, no phenotypic or genetic data pertinent to these areas are available. While there were clear ecoregional differences and large distances between the uppermost and lowermost populations in the Snake basin, there was no clear point of division between the two areas. Rather, populations and MPGs in the middle Snake (e.g., Payette, Boise, and Malheur rivers) had mosaic characteristics of both upper and lower areas and could have provided potential connectivity. Faced with clear differences between upper and lower regions, but without a clear point at which to divide ESUs, we did not delineate an extirpated ESU in this region. Rather, we maintained the dual possibility that historically there may have been one, extremely large, continuous ESU, or that there may have been multiple ESUs in the Snake Basin.

Table A-1. Summary of potential contributions to ESU function by extirpated MPGs in the Snake River spring/summer chinook ESU. Two plus marks “++” indicates that the MPG would play a relatively large role in the ESU for this characteristic. A single plus mark “+” indicates that the MPG would play a relatively smaller role in the ESU for this characteristic, or that several MPGs would be required for the benefit to be realized

MPG	Habitat Quantity	Spatial Structure	Diversity
Dry Clearwater (lower)	+	++	++
Wet Clearwater (upper)	+	+	++
Middle Snake (Pine to Weiser)	+	+	+
Payette/Boise	+	+	+
Malheur	+	+	++
Owyhee	+	+	++
Upper Snake (Snake tribs to Rock Cr.)	+	+	++

Abundance and Productivity – Habitat Quantity

In total, an area equaling more than twice the currently accessible area has been extirpated from the Snake River spring/summer chinook ESU (Table A-2). However, currently accessible area includes more than 2,000 kilometers of habitat (kilometers weighted by quality). Thus, while the inclusion of any additional MPG, particularly some of the larger MPGs (e.g. Payette/Boise or Malheur) would substantially increase available habitat, we did not feel that tributary habitat quantity (as a surrogate for ESU abundance and productivity) was limiting ESU viability.

Table A-2. Habitat quantity in extant and extirpated MPGs of the Snake River spring/summer Chinook ESU. Quantity is reported in weighted kilometers, with areas of "high" intrinsic potential receiving a weight of 1; moderate receiving a weight of 0.5, and low areas receiving a weight of 0.25. *Weighted kilometers of extant MPGs include any extirpated populations.

MPG	Weighted stream km	% of Extant ESU	% of Total ESU
EXTANT			
Lower Snake River	124.5	5.98	1.90
Grande Ronde / Imnaha*	526.4	25.28	8.02
South Fork Salmon River	232.6	11.17	3.54
Middle Fork Salmon River	422.5	20.29	6.43
Upper Salmon River*	775.9	37.27	11.82
Extant MPGs Total	*2081.9*	*100.00*	*31.71*
EXTIRPATED			
Dry Clearwater (lower)	318.60	15.30	4.85
Wet Clearwater (upper)	588.90	28.29	8.97
Middle Snake (Pine to Weiser)	628.74	30.20	9.58
Payette/Boise	819.65	39.37	12.48
Malheur	533.29	25.62	8.12
Owyhee	818.38	39.31	12.46
Upper Snake (Snake tributaries to Rock Cr.)	776.22	37.28	11.82
Extirpated MPGs Total	*4483.78*	*215.37*	*68.29*
Total ESU	***6565.68***	***315.37***	***100.00***

Connectivity -- Spatial Structure

Most of the area from which Snake River spring/summer chinook have been extirpated is in the most upstream areas of the potential range. However, extirpation from the Clearwater River resulted in a gap in connectivity between currently extant MPGs. The Lower Snake MPG, in particular, is currently more isolated from other components of the ESU than was likely historically (Table A-3).

Table. A-3. Distance between extant and extirpated MPGs and the closest neighboring MPGs in the Snake River spring/summer chinook ESU under two conditions: 1) that only extant MPGs are occupied; and 2) that all MPGs are occupied. Distance measured from the most downstream area rated "moderate" in the IC-TRT's intrinsic potential analysis.

MPG	Closest Currently Occupied MPG	Distance (km)	Closest Historically Occupied MPG	Distance (km)	Difference in Distance
EXTANT					
Lower Snake River	Grande Ronde/ Imnaha	114.93	Dry Clearwater	30.79	84.14
Grande Ronde / Imnaha	Lower Snake	114.93	Lower Snake	114.93	0
South Fork Salmon River	Grande Ronde/ Imnaha	131.44	Grande Ronde/ Imnaha	131.44	0
Middle Fork Salmon River	Upper Salmon	38.13	Upper Salmon	38.13	0
Upper Salmon River	Middle Fork Salmon	38.13	Middle Fork Salmon	38.13	0
EXTIRPATED					
Dry Clearwater (lower)	Lower Snake	30.79	Lower Snake	30.79	0
Wet Clearwater (upper)	Lower Snake	85.79	Dry Clearwater	16.61	69.18
Middle Snake (Pine to Weiser)	Grande Ronde/ Imnaha	180.28	Grande Ronde/ Imnaha	180.28	0
Payette/Boise	Grande Ronde/ Imnaha	339.69	Malheur	19.88	319.81
Malheur	Grande Ronde/ Imnaha	331.31	Payette/Boise	19.88	311.43
Owyhee	Grande Ronde/ Imnaha	434.76	Upper Snake	96.78	337.98
Upper Snake (Snake tributaries to Rock Cr.)	Grande Ronde/ Imnaha	406.85	Owyhee	96.78	310.07

Habitat types – Diversity

All extirpated MPGs include a substantial amount of area in ecoregions different from those represented by extant MPGs (Table A-4). Therefore, we anticipate that all of these MPGs likely contributed to the phenotypic diversity expressed within the ESU with greatest potential contribution from Clearwater, Malheur, Owyhee, and Upper Snake MPGs. Thus, re-population of the upper reaches could contribute substantially to either basin-wide diversity as separate ESUs or within-ESU diversity as separate MPGs. Repopulation of the middle reaches would likely result in smaller increases in diversity.

Table A-4. Distribution (percentage) of extant and extirpated MPGs in the Snake River spring/summer chinook ESU across EPA ecoregions (level 3). Areas rated “moderate” and “high” in the IC-TRT’s intrinsic potential analysis were included in this estimate.

MPG	**Blue Mountains**	**Columbia Plateau**	**Idaho Batholith**	**Middle Rockies**	**Northern Basin and Range**	**Northern Rockies**	**Snake River Plain**
EXTANT							
Grande Ronde / Imnaha	100.0						
Lower Snake	8.8	91.2					
South Fork Salmon River	6.0		94.0				
Middle Fork Salmon River			100.0				
Upper Salmon River			44.6	55.4			
Extant MPGs Total	25.5	5.3	47.0	22.3			
EXTIRPATED							
Dry Clearwater (lower)		14.6	37.0			48.4	
Wet Clearwater (upper)			39.4			60.6	
Middle Snake (Pine to Weiser)	67.4		0.3				32.3
Payette/Boise			73.9				26.1
Malheur	24.6				49.3		26.1
Owyhee					98.6		1.4
Upper Snake (Snake tributaries to Rock Cr.)			1.5		49.5		49.0
Extirpated MPGs Total	7.1	0.5	20.4		42.2	4.2	25.7
Total ESU	**10.2**	**1.3**	**24.9**	**3.8**	**35.0**	**3.5**	**21.3**

B. Snake River steelhead

None of the extirpated MPGs in the Snake River steelhead ESU would likely substantially increase the probability of long-term persistence of this ESU. Although the extirpated MPGs would contribute to the quantity and diversity of habitats available to the ESU, particularly those in the upper portion (Table B-1), there is currently a large amount of habitat available to the ESU, spread across several MPGs and ecoregions. This ESU is unique in having a small portion of a single population within an extirpated MPG (Hells Canyon) that is still accessible to anadromous fish. If fish in this area are descended from one or more historical populations, maintaining this genetic legacy would contribute to overall ESU diversity.

We evaluated the possibility that there might have been one or more ESUs above the current Hells Canyon dam complex historically (see Population Identification update memo; further discussion to be provided in final Population Identification document). Unfortunately, no phenotypic data pertinent to these areas are available; currently available genetic data on resident redband trout were not illuminating, and may not be relevant for the anadromous life history. While there were clear ecoregional differences and large distances between the uppermost and lowermost populations in the Snake basin, there was no clear point of division between the two areas. Rather, populations and MPGs in the middle Snake (e.g. Payette, Boise, and Malheur Rivers) had mosaic characteristics of both upper and lower areas and could have provided potential connectivity. Faced with clear differences between upper and lower regions, but without a clear point at which to divide ESUs, we did not delineate an extirpated ESU in this region. Rather, as with the Snake River spring/summer Chinook ESU, we maintained the dual possibility that historically there may have been one, extremely large, continuous ESU, or that there may have been multiple ESUs in the Snake Basin.

Table B-1. Summary of potential contributions to ESU function by extirpated MPGs in the Snake River steelhead ESU. Two plus marks "++" indicates that the MPG would play a relatively large role in the ESU for this characteristic. One plus mark "+" indicates that the MPG would play a relatively smaller role in the ESU for this characteristic, or that several MPGs would be required for the benefit to be realized.

MPG	Habitat Quantity	Spatial Structure	Diversity
Hells Canyon*	+	+	+
Payette/Boise	+	+	+
Malheur/Owyhee	+	+	++
Bruneau and Salmon Falls	+	+	++

* Several small tributaries in the lower reaches of Hells Canyon are currently occupied by steelhead. However, this is an extremely small component of the entire MPG; we thus treat this MPG as an extirpated area for calculation of comparison statistics.

Habitat Quantity – Abundance and Productivity

Extirpated areas in the Snake River steelhead ESU are approximately equal to the areas currently occupied (Table B-2). However, currently there are more than 12,000 stream km (weighted by intrinsic quality) available to this ESU. Thus, we did not consider habitat quantity (as a surrogate for abundance and productivity) to be impairing the viability of this ESU.

Table B-2. Habitat quantity in extant and extirpated MPGs of the Snake River steelhead ESU. Quantity is reported in weighted kilometers, with areas of "high" intrinsic potential receiving a weight of 1; moderate receiving a weight of 0.5, and low areas receiving a weight of 0.25. *Weighted kilometers of extant MPGs include any extirpated populations.

MPG	Weighted stream km	% of extant ESU	% of total ESU
EXTANT			
Lower Snake	834.16	6.91	3.24
Clearwater River	3757.26	31.11	14.59
Grande Ronde River	2259.92	18.71	8.77
Salmon River	4760.29	39.42	18.48
Imnaha River	465.58	3.86	1.81
Extant MPGs Total	*12077.21*	*100.00*	*46.89*
EXTIRPATED			
Hells Canyon*	3,193.17	26.44	12.40
Payette/Boise	3,236.94	26.80	12.57
Malheur/Owyhee	4,348.90	36.01	16.89
Bruneau and Salmon Falls	2,898.40	24.00	11.25
Extirpated MPGs Total	*13,677.41*	*113.25*	*53.11*
Total ESU	***25,754.62***	***213.25***	***100.00***

* Several small tributaries in the lower reaches of Hells Canyon are currently occupied by steelhead. However, this is an extremely small component of the entire MPG; we thus treat this MPG as an extirpated area for calculation of comparison statistics.

Connectivity – Spatial Structure

None of the extirpated MPGs alone impair the connectivity of extant MPGs (Table B-3). Extirpated MPGs in this ESU are all congruent, and located in the upstream portion of the potential range. However, if areas currently occupied in the Hells Canyon MPG contain remnants of historical populations, extirpated areas of that MPG would be important to the MPG spatial structure in light of the limited distribution and size of extant populations.

Table B-3. Distance between extant and extirpated MPGs and the closest neighboring MPGs under two conditions: 1) that only extant MPGs are occupied; and 2) that all MPGs are occupied. Distance measured from the most downstream area rated "moderate" in the IC-TRT's intrinsic potential analysis.

MPG	Closest Currently Occupied MPG	Distance (km)	Closest Historically Occupied MPG	Distance (km)	Difference in Distance
EXTANT					
Lower Snake	Grande Ronde	2.68	Grande Ronde	2.68	0
Clearwater River	Grande Ronde	12.23	Grande Ronde	12.23	0
Grande Ronde River	Lower Snake	2.68	Lower Snake	2.68	0
Salmon River	Imnaha River	29.06	Imnaha River	29.06	0
Imnaha River	Hells Canyon	3.00	Hells Canyon	3.00	0
EXTIRPATED					
Hells Canyon*	Imnaha River	129.49	Imnaha River	129.49	0
Payette/Boise	Imnaha River	304.49	Malheur/Owyhee	45.34	259.15
Malheur/Owyhee	Imnaha River	318.73	Bruneau and Salmon Falls	31.16	287.57
Bruneau and Salmon Falls	Imnaha River	355.52	Malheur/Owyhee	31.16	324.36

Habitat Types – Diversity

All of the extirpated MPGs, if occupied, would expand the range of ecoregions encountered by fish in this ESU (Table B-4). However, the currently occupied areas cover five ecoregions. Re-population of the upper reaches could contribute substantially to either basin-wide diversity as separate ESUs or within-ESU diversity as separate MPGs. Repopulation of the middle reaches would likely result in smaller increases in diversity.

Importantly, fish that currently occupy the remaining accessible small tributaries of the Hells Canyon MPG may be the only remnants of upstream populations. Alternatively, they may be strays from hatchery programs. Emphasis should be placed on determining the origin of these fish. If they do appear to be remnants of an historical population, maintaining these fish would preserve this genetic legacy.

Table B-4. Distribution (percentage) of extant and extirpated MPGs in the Snake River steelhead ESU across EPA ecoregions (level 3). Areas rated "moderate" and "high" in the IC-TRT's intrinsic potential analysis were included in this estimate.

MPG	**Blue Mountains**	**Columbia Plateau**	**Idaho Batholith**	**Middle Rockies**	**Northern Basin and Range**	**Northern Rockies**	**Snake River Plain**
EXTANT							
Lower Snake	17.8	82.2					
Clearwater River	0.3	5.2	43.7			50.9	
Grande Ronde River	97.5	2.5					
Salmon River	4.7	0.3	72.0	23.1			
Extant MPGs Total	23.1	8.6	42.8	9.2		16.3	
EXTIRPATED							
Hells Canyon	67.2		0.3				32.4
Payette/Boise			75.0				25.0
Malheur/Owyhee	4.5				92.6		2.9
Bruneau and Salmon Falls			1.5		49.6		48.9
Extirpated MPGs Total	5.8		19.0		48.1		27.1
Total ESU	**16.0**	**5.1**	**33.1**	**5.5**	**19.7**	**9.6**	**11.1**

C. Snake River fall chinook

We include all three Snake River fall chinook populations in a single MPG. This ESU does not include any extirpated MPGs. However, the single MPG must be at low risk for the ESU to be considered viable. This would require the re-establishment of at least one population to meet the minimum viability criteria we have established. We recognize that there are significant difficulties in re-establishing fall chinook populations above the Hells Canyon complex, and suggest that initial effort be placed on recovery for the extant population, concurrently with scoping efforts for re-introduction, as described above in the adaptive approach.

D. Snake River sockeye

We do not have data to support an intrinsic potential analysis for Snake River sockeye. Lakes or groups of lakes that formerly supported sockeye salmon in the Snake River drainage are: Wallowa Lake, Payette Lake basin, and Warm Lake. However, each of these lake groups is separated by distances that are consistent with those between other sockeye ESUs. It is unclear, and currently irresolvable, whether these lake groups were MPGs of the same ESU or separate ESUs. Thus, re-population of these additional lake basins could contribute substantially to either basin-wide diversity as separate ESUs or within-ESU diversity as separate MPGs. Ultimately, three populations within the Stanley Lakes Basin, however, will be required for this ESU to meet minimum ESU viability criteria. This issue is treated in greater detail in our MPG-ESU scenarios memo.

E. Upper Columbia spring chinook

The repopulation of either the Spokane or the Kettle/Colville/San Poil MPG would substantially reduce the overall risk faced by the Upper Columbia spring chinook ESU. This judgment was based on the combination of likely contribution to overall ESU abundance and productivity, diversity and spatial structure (Table E-1), given the small number and extent, potential for catastrophic loss, and low diversity of the single extant MPG.

Table E-1. Summary of potential contributions to ESU function by extirpated MPGs in the Upper Columbia spring chinook ESU. Two plus marks "++" indicates that the MPG would play a relatively large role in the ESU for this characteristic. One plus mark "+" indicates that the MPG would play a relatively smaller role in the ESU for this characteristic, or that several MPGs would be required for the benefit to be realized.

MPG	Habitat Quantity	Spatial Structure	Diversity
Kettle/Colville/ San Poil	++	++	++
Spokane	++	++	++

Habitat Quantity – Abundance and Productivity

While the currently occupied East Cascades MPG is the largest MPG in this ESU, the total currently accessible habitat is relatively low (less than 700 weighted stream km) (Table E-2). Some of the area noted within the extirpated areas may have been occupied by summer chinook, which are a different ESU. However, either of the extirpated MPGs would contribute substantially to the total amount of available habitat. If both were occupied, habitat quantity could as much as double.

Table E-2. Habitat quantity in extant and extirpated MPGs of the Upper Columbia River spring Chinook ESU. Quantity is reported in weighted kilometers, with areas of "high" intrinsic potential receiving a weight of 1; moderate receiving a weight of 0.5, and low areas receiving a weight of 0.25. *Weighted kilometers of extant MPGs include any extirpated populations.

MPG	Weighted stream km	% of extant ESU	% of total ESU
EXTANT			
East Cascades*	640.1	100.00	43.62
Extant MPGs Total	*640.1*	*100.00*	*43.62*
EXTIRPATED			
Kettle/Colville/San Poil	443.1	69.22	30.19
Spokane	384.27	60.03	26.19
Extirpated MPGs Total	*827.37*	*129.26*	*56.38*
Total ESU	***1467.47***	***229.26***	***100.00***

Connectivity – Spatial Structure

Neither extirpated MPG would contribute substantially to connectivity of the single MPG. However, the presence of either would alleviate the likelihood of a common catastrophe or other spatially-linked impact affecting the entire ESU.

Table E-3. Distance between extant and extirpated MPGs and the closest neighboring MPGs under two conditions: 1) that only extant MPGs are occupied; and 2) that all MPGs are occupied. Distance measured from the most downstream area rated "moderate" in the IC-TRT's intrinsic potential analysis.

MPG	Closest Currently Occupied MPG	Distance (km)	Closest Historically Occupied MPG	Distance (km)	Difference in Distance
EXTANT					
East Cascades	none		Kettle/Colville/San Poil	182.52	182.52
EXTIRPATED					
Kettle/Colville/San Poil	East Cascades	182.52	Spokane	31.99	150.53
Spokane	East Cascades	214.36	Kettle/Colville/San Poil	31.99	182.37

Habitat Types – Diversity

Both extirpated MPGs occur in ecoregions that are different from those in the currently accessible MPG. Access to these areas would likely increase the potential for a greater range of phenotypic diversity within the ESU.

Table E-4. Distribution (percentage) of extant and extirpated MPGs in the Upper Columbia spring chinook ESU across EPA ecoregions (level 3). Areas rated "moderate" and "high" in the IC-TRT's intrinsic potential analysis were included in this estimate

MPG	**Columbia Plateau**	**North Cascades**	**Northern Rockies**
EXTANT			
Below Chief Joseph Dam	26.1	73.9	0.1
EXTIRPATED			
Kettle/Colville/San Poil	1.9		98.1
Spokane	48.1		51.9
Extirpated MPGs Total	33.1		66.9
Total ESU	**30.5**	**27.2**	**42.2**

F. Upper Columbia steelhead

Repopulation of either the Spokane or the Kettle/Colville/San Poil MPG would substantially reduce the risk of the Upper Columbia steelhead ESU. This judgment was based on the combination of likely contribution to overall ESU abundance and productivity, diversity and spatial structure (Table F-1). This situation and our rationale are similar to that for Upper Columbia Spring Chinook.

Table F-1. Summary of potential contributions to ESU function by extirpated MPGs in the Upper Columbia steelhead ESU. Two plus marks "++" indicates that the MPG would play a relatively large role in the ESU for this characteristic. One plus mark "+" indicates that the MPG would play a relatively smaller role in the ESU for this characteristic, or that several MPGs would be required for the benefit to be realized.

MPG	Habitat Quantity	Spatial Structure	Diversity
Kettle/Colville/ San Poil	++	++	++
Spokane	++	++	++

Habitat Quantity – Abundance and Productivity

Currently accessible habitat for steelhead in the Upper Columbia total approximately 3500 weighted kilometers (Table F-2). However, the MPG with the largest potential habitat quantity in this ESU is currently extirpated.

Table F-2. Habitat quantity in extant and extirpated MPGs of the Upper Columbia River steelhead ESU. Quantity is reported in weighted kilometers, with areas of "high" intrinsic potential receiving a weight of 1; moderate receiving a weight of 0.5, and low areas receiving a weight of 0.25. *Weighted kilometers of extant MPGs include any extirpated populations.

MPG	Weighted stream km	% of extant ESU	% of total ESU
EXTANT			
East Cascades*	3527.55	100.00	40.98
Extant MPGs Total	*3527.55*	*100.00*	*40.98*
EXTIRPATED			
Kettle/Colville/San Poil River	4,009.35	113.66	46.58
Spokane River	1,070.32	30.34	12.44
Extirpated MPGs Total	*5,079.67*	*144.00*	*59.02*

Total ESU	***5,079.67***	***244.00***	***100.00***

Connectivity – Spatial Structure

Neither extirpated MPG would contribute substantially to connectivity of the single MPG (Table F-3). However, the presence of either would alleviate the likelihood of a common catastrophe or other spatially-linked impact affecting the entire ESU.

Table F-3. Distance between extant and extirpated MPGs and the closest neighboring MPGs under two conditions: 1) that only extant MPGs are occupied; and 2) that all MPGs are occupied. Distance measured from the most downstream area rated "moderate" in the IC-TRT's intrinsic potential analysis.

MPG	Closest Currently Occupied MPG	Distance (km)	Closest Historically Occupied MPG	Distance (km)	Difference in Distance
EXTANT					
East Cascades	None		Kettle/Colville/San Poil	49.44	
EXTIRPATED					
Kettle/Colville/San Poil River	East Cascades	49.44	Spokane	19.87	29.57
Spokane River	East Cascades	199.88	Kettle/Colville/San Poil	19.87	180.01

Habitat Types – Diversity

Both extirpated MPGs occur in ecoregions that are different from those encountered by fish in the currently accessible MPG. Access to these areas would likely increase the potential for a greater range of phenotypic diversity within the ESU.

Table F-4. Distribution (percentage) of extant and extirpated MPGs in the Upper Columbia steelhead ESU across EPA ecoregions (level 3). Areas rated "moderate" and "high" in the IC-TRT's intrinsic potential analysis were included in this estimate.

MPG	**Columbia Plateau**	**Eastern Cascades Slopes and Foothills**	**North Cascades**	**Northern Rockies**
EXTANT				
East Cascades - Below Chief Joseph Dam	40.0	0.4	58.8	0.9
EXTIRPATED				
Kettle/Colville/San Poil River	0.9			99.1
Spokane River	42.5			57.5
Extirpated MPGs Total	19.7			80.3
Total ESU	**26.9**	**0.1**	**21.0**	**52.0**

G. Mid-Columbia steelhead

No MPGs in the Mid-Columbia steelhead ESU have been completely extirpated. Extirpated populations and subpopulations within this ESU should be considered within the context of MPG viability. We treat these areas in greater detail in our MPG-ESU scenario memo.

References

Ruckelshaus, M., P. McElhany, M. McClure and S. Heppell. 2004. Chinook salmon in Puget Sound: Effects of spatially correlated catastrophes on persistence. Pp. 208-218, In R. Ackakaya, M. Burgman, O. Kindvall, C.C. Wood, P. Sjogren-Gulve, J. S. Hatfield and M. A. McCarthy (eds.) Species Conservation and Management: Case Studies. Oxford Univ. Press.

UNITED STATES DEPARTMENT OF COMMERCE
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NATIONAL MARINE FISHERIES SERVICE
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MEMORANDUM

Date: January 8, 2007
From: Interior Columbia Technical Recovery Team
To: NMFS NW Regional Office, co-managers and other interested parties
Subject: Scenarios for MPG and ESU viability consistent with TRT viability criteria

Purpose and Scope

Clearly, the overall goal of recovery planning is to achieve a condition for an ESU where it no longer needs protection under the ESA because it is no longer in danger of extinction or likely to become endangered within the foreseeable future. The ICTRT (2005, 2006) viability criteria recommend that all Major Population Groups (MPGs) in the ESU must be viable before the ESU can be considered at low risk of extinction and a candidate for delisting. Because of the importance of the MPG in determining overall ESU viability, we are providing more focused interpretation and application of ICTRT MPG-level viability criteria. In this memo, we provide, for each MPG in the Interior Columbia recovery domain, a discussion about the combinations of populations that would meet the ICTRT MPG-level recovery criteria if those populations achieved low risk status. We also provide some recommendations and considerations that recovery planners could use to prioritize populations for meeting viability criteria within an MPG. However, in most cases where there are multiple possible combinations of populations that could achieve MPG and ESU viability, we do not provide a single set of populations. Likewise, we did not develop a "least-effort" scenario for achieving MPG viability. While we considered providing such a population set, we concluded there were multiple ways to identify a "least-effort" scenario technically and that scenario would also involve social, economical, and political considerations that are outside of our purview. We do provide some discussion about ways in which populations could be prioritized for recovery efforts.

The "TRT-recommendation" included in this memo for each MPG is a description of populations that, when those populations achieve viable status, would meet the minimum MPG-level viability criteria. The populations included in each recommendation or viable-MPG scenario were selected based on unique characteristics (e.g. run timing, populations size, genetic characteristics), major production areas in the MPG, and spatial distribution of the populations. Importantly, although not all populations in a MPG need to meet TRT viability criteria under most viable-MPG scenarios, it is strongly advisable to attempt to improve the status of more than the minimum number of populations to a low-risk (viable) situation. There are two primary reasons for this:

First, based on current population dynamic theory, the TRT has recommended that all extant populations be maintained with sufficient productivity that the overall MPG productivity does not fall below replacement (i.e. these areas should not serve as significant population sinks). Thus, it would be highly risky to allow the status of any population to degrade. In fact, many populations will need to be improved from their current status to be regarded as "maintained." As a rule of thumb, the TRT believes that populations that fall within cells adjacent to those that we regard as viable in our risk matrix (Figure 1) can be regarded as "maintained." We will provide further discussion of this issue in a forthcoming update to our viability document.

Figure 1. Matrix of possible Abundance/Productivity and Spatial structure/Diversity scores for application at the population level. Percentages for abundance and productivity (A/P) scores represent the probability of extinction in a 100-year time period. Cells that contain a "V" are considered viable combinations; "HV" indicates Highly Viable combinations; "M*" indicates combinations that can be regarded as candidates for "maintained." The darkest cells represent combinations of A/P and SSD at greatest risk.

		Spatial Structure/Diversity Risk			
		Very Low	**Low**	**Moderate**	**High**
Abundance/ Productivity Risk	**Very Low** (<1%)	**HV**	**HV**	**V**	M*
	Low (1-5%)	**V**	**V**	**V**	M*
	Moderate (6 – 25%)	M*	M*	M*	
	High (>25%)				

Second, although the possible population sets suggested in this memo would meet TRT recovery criteria for the ESUs, achieving recovery for those populations will likely require attempting recovery in more than just those populations because of the uncertainty of success of recovery efforts. For example, if there is an 80% chance that recovery will be successful in each of a set of three populations identified, there is an overall 51% probability of recovering three populations if recovery efforts are limited to those three populations (McElhany et al. 2003). To have more than a 95% probability of recovering three populations in this case would require attempting recovery of six populations. <u>A low-risk strategy will thus target more populations than the minimum for viability.</u>

<u>Prioritizing Populations within Scenarios</u>

Prioritizing populations is by its nature, a technical and policy exercise. In this memo, we provide descriptions of scenarios that would meet TRT biological viability criteria.

Because these are not, in most cases, a single scenario, we also identified a number of additional factors that could be considered by recovery planners choosing which populations to target in order to meet MPG viability criteria:

- Current status of the population – Recovery planners should consider the current condition of the population with respect to all four VSP parameters. Those that are closest to viability criteria currently may require less effort (but the remaining factors should also be considered.)
- Biological feasibility – This is closely tied to the current status of the population, but includes considerations, for example, of whether particular actions can produce the needed change. It also includes considerations for density-dependence –for example, would the required change be feasible, given current spawner or juvenile capacity?
- Political/social/economic feasibility – Obviously, some recovery actions are constrained by non-biological factors. These may make a population less or more attractive to serve as a low-risk/viable population than it would be by strictly biological criteria.
- Hatchery practices affecting the population – hatchery practices and diversity criteria in some locations may be in conflict. This may affect the choice of populations.
- Monitoring history – Some populations have an extensive history of monitoring data, while others have very little. It may cost less in dollars and effort to determine that a population has met viability criteria with substantial existing monitoring data.
- Presence of multiple species in an area that would benefit by the same actions. Populations may rise in importance when more than one species of concern is in the area, and actions would achieve efficiencies of effort.

<u>MPG-level scenarios consistent with TRT criteria for each ESU</u>

To achieve viable ESUs in the Interior Columbia, the TRT recommends that all extant MPGs meet MPG-level criteria. We, therefore, present combinations of populations within MPGs that would meet viability criteria. For each MPG, we first present the "menu" of populations that would meet our criteria. We then discuss population-specific characteristics or conditions that should be considered when choosing among populations in that menu. Finally, we provide a reduced set of populations that we recommend meet our criteria. We will be providing additional information about the IC-TRT's recommended approaches to MPGs that include populations that have been extirpated in another memo.

In this document, we identify recommendations and scenarios that are consistent with our criteria.

A. Snake River spring/summer chinook salmon ESU

For the Snake River spring/summer chinook salmon ESU to meet TRT viability criteria, each of the MPGs should meet the scenarios described below:

1. Lower Snake MPG

Component populations:

	Size Category	Life History Type
Tucannon River	Intermediate	Spring
Asotin Creek (functionally extirpated)	Basic	Spring

Basic application of TRT criteria:

- Two populations must meet viability criteria, one of which must meet high viability criteria

Considerations:

- Asotin Creek population is functionally extirpated. Treatment of extirpated populations is discussed more thoroughly in the accompanying memo. However, our general recommendation is that extirpated populations be included in the total number of populations in the ESU (for calculating minimum number of populations in the MPG), but that the initial focus of recovery efforts be put on extant populations, with scoping efforts for re-introductions conducted concurrently.

TRT Recommendation:

Highly Viable:	Tucannon River (receives initial focus)
Re-considered for reintroduction as recovery efforts progress:	Asotin Creek

2. Grande Ronde/Imnaha MPG

Component populations:

	Size Category	Life History Type
Wenaha River	Intermediate	Spring
Minam River	Intermediate	Spring
Lostine/Wallowa Rivers	Large	Spring
Lookingglass Creek (functionally extirpated)	Basic	Spring
Catherine Creek	Large	Spring
Upper Grande Ronde	Large	Spring
Imnaha River	Intermediate	Spring/Sum
Big Sheep Creek (functionally extirpated)	Basic	Spring

Basic application of TRT criteria:

- Four populations must meet viability criteria, one of which must meet high viability criteria
- Population in the Imnaha River has a unique life history strategy; this must meet viability criteria
- Two of the three Large populations must meet viability criteria

Considerations:

- Lookingglass Creek and Big Sheep Creek populations are functionally extirpated.
- Distributing viable "Large" populations throughout the sub-basin is preferable to having them clumped or contiguous.
- There is the potential for Imnaha to be isolated.
- Wenaha R. is most downstream, providing connectivity with other MPGs.
- Wenaha R. and Minam R. populations are currently the most unaffected by hatchery fish. Hatchery supplementation programs are ongoing in the Imnaha, Wallowa-Lostine, Catherine Creek and Upper Grande Ronde populations.
- Minam R. and Wenaha R. populations have little spatial structure or diversity impairment. They may be candidates for high viability status.

TRT Recommendation:

1 Highly Viable and 3 Viable:	Imnaha River Lostine/Wallowa River Catherine Creek OR Upper Grande Ronde R. Wenaha R. OR Minam R.
Maintained:	All remaining extant populations

3. South Fork Salmon MPG

Component populations:

	Size Category	Life History Type
Little Salmon River (includes Rapid River)	Intermediate	Spring/Sum
South Fork Salmon River	Large	Summer
Secesh River	Intermediate	Summer
East Fork South Fork Salmon River	Large	Summer

Basic application of TRT criteria:

- Two populations minimum must meet viability criteria, one of which must meet high viability criteria.
- Little Salmon River (as the only spring/summer life history).
- One Large population (East Fork South Fork or South Fork) must meet viability criteria.

Considerations:

- The Little Salmon's size category is largely driven by small, adjunct tributaries. These adjunct tributaries are also the only places where the spring life history is represented in the population. If this was not the case historically (i.e. if these fish are a result of hatchery production or not representative of the historical condition), the importance of maintaining that life history is somewhat less.
- Little Salmon River population is greatly influenced by Rapid River hatchery production and releases.
- Ongoing supplementation exists in EFSF population (Johnson Creek).

TRT Recommendation:

1 Highly Viable and 1 Viable:	Two populations in the main South Fork basin.
Maintained:	All remaining extant populations

4. Middle Fork Salmon MPG

Component populations:

	Size Category	Life History Type
Middle Fork Salmon below Indian Creek	Basic	Spring/Sum
Big Creek	Large	Spring/Sum
Camas Creek	Basic	Spring
Loon Creek	Basic	Spring/Sum
Middle Fork Salmon above Indian Creek	Intermediate	Spring
Sulphur Creek	Basic	Spring
Bear Valley/Elk Creek	Intermediate	Spring
Marsh Creek	Basic	Spring
Chamberlain Creek	Intermediate	Spring

Basic application of TRT criteria:

- Five populations must meet viability criteria, one of which must meet high viability criteria.
- Big Creek is required by size criteria.
- Two of three Intermediate populations (Middle Fork Salmon above Indian Creek, Chamberlain Creek, or Bear Valley Creek) must meet viability criteria, to meet size criteria.

Considerations:

- Chamberlain Creek falls in a significant geographic position – providing connectivity between MPGs.
- Chamberlain Creek has unique, apparently persistent genetic characteristics.
- Marsh Creek is somewhat less isolated, and overall a larger production area than Sulphur Creek.
- Upper Middle Fork mainstem is composed of a number of small tributaries (rather than a core, contiguous spawning area).
- Several populations have potential to achieve Highly Viable status because of high quality habitat.

TRT Recommendation:

1 Highly Viable and 4 Viable:	Big Creek Chamberlain Creek Bear Valley Creek Marsh Creek Camas OR Loon Creek
Maintained:	All remaining extant populations

5. Upper Salmon MPG

Component populations:

	Size Category	Life History Type
North Fork Salmon River	Basic	Spring
Panther Creek (extirpated)	Intermediate	Spring
Lemhi River	Very Large	Spring
Salmon River mainstem, below Redfish Lake	Very Large	Spring/Sum
Pahsimeroi River	Large	Spring
East Fork Salmon River	Large	Spring/Sum
Yankee Fork	Basic	Spring
Valley Creek	Basic	Spring
Upper Salmon River mainstem, above Redfish Lake	Large	Spring

Basic application of TRT criteria:

- Five populations must meet viability criteria, one of which must meet high viability criteria
- Pahsimeroi River has the only extant summer life history strategy, and thus must meet viability criteria
- Three Very Large or Large populations (Lemhi R., Pahsimeroi, East Fork Salmon R., Salmon River mainstem, above and below Redfish Lake) must meet viability criteria
- One Intermediate or larger population (Panther Creek is the only Intermediate population) must meet viability criteria.

Considerations:

- Lemhi historically may have had summer chinook production.
- Panther Creek is extirpated and is the only intermediate population; a large population could be substituted for it.
- Lemhi provides important connectivity to other MPGs, as a large, downstream population.
- Upper Salmon mainstem population is at the geographic “end” of the ESU and MPG.
- Valley Creek had historically larger production than most Basic populations.
- North Fork is the most downstream population. However, fairly few data are available, and substantial anthropogenic effects to population and habitat.
- Yankee Fork is currently occupied by non-native stock.

TRT Recommendation:

1 Highly Viable and 4 Viable:	Lemhi R. Pahsimeroi R. East Fork Salmon River Upper Salmon River Valley Creek
Maintained:	All remaining extant populations

B. Snake River Steelhead DPS

1. Lower Snake MPG

Component populations:

	Size Category	Life History Type
Tucannon River	Intermediate	A-Run
Asotin Creek	Basic	A-Run

Basic application of TRT criteria:

- Two populations must meet viability criteria, one of which must meet high viability criteria

Considerations: (none)

TRT Recommendation:

1 Highly Viable and 1 Viable:	Tucannon River Asotin Creek

2. Clearwater MPG

Component populations:

	Size Category	Life History Type
Lower Clearwater	Large	A-Run
South Fork Clearwater	Intermediate	B-Run
North Fork Clearwater (extirpated)	Large	
Lolo Creek	Basic	A&B-Run
Selway River	Intermediate	B-Run
Lochsa River	Intermediate	B-Run

Basic application of TRT criteria:
- Three populations must meet viability criteria, one of which must meet high viability criteria.
- Lolo Creek has the only A and B life history, and must meet viability criteria.
- Two Large or Very Large populations (North Fork Clearwater, Lower Clearwater, Lochsa or Selway) must meet viability criteria.
- One additional Intermediate or larger population must meet viability criteria.
- At least one A-run and one B-run population must meet viability criteria.

Considerations:
- TRT criteria for size and life history cannot be met with three populations; four are necessary.
- Lochsa River is more accessible than the Selway River for data collection.
- North Fork population is extirpated.
- A/B life history (as seen in Lolo) may be less important than ensuring that both A-run and B-run fish are present.

TRT Recommendation:

1 Highly Viable and 3 Viable:	Lower Clearwater Lolo Creek 2 of: Selway River, Lochsa River, SF Clearwater
Maintained:	All remaining extant populations

3. Grande Ronde MPG

Component populations:

	Size Category	Life History Type
Lower Grande Ronde mainstem	Intermediate	A-Run
Joseph Creek	Basic	A-Run
Wallowa River	Intermediate	A-Run
Upper Grande Ronde mainstem	Large	A-Run

Basic application of TRT criteria:
- Two populations must meet viability criteria, one of which must meet high viability criteria.
- Grande Ronde upper mainstem must meet viability criteria, as the only Large population.

Considerations:
- The Lower mainstem or Joseph populations would contribute to spatial structure in the lower portion of the MPG.
- Wallowa includes multiple core areas, some unique habitat characteristics (Eagle Caps), but does support a hatchery (with little straying).
- Joseph Creek population is somewhat smaller than the others and has the least hatchery influence.
- Lower Grande Ronde population receives hatchery releases.
- Upper Grande Ronde population currently receives no hatchery releases.
- Joseph Creek may be a candidate for High Viability status.

TRT Recommendation:

1 Highly Viable and 1 Viable:	Upper Grande Ronde Joseph Creek OR Lower Grande Ronde
Maintained:	All remaining extant populations

4. Salmon River MPG

Component populations:

	Size Category	Life History Type
Little Salmon and Rapid Rivers	Intermediate	A-Run
South Fork Salmon River	Intermediate	B-Run
Secesh River	Basic	B-Run
Lower Middle Fork Tributaries	Large	B-Run
Upper Middle Fork Salmon River	Large	B-Run
Chamberlain Creek	Basic	A-Run
Panther Creek	Basic	A-Run
North Fork Salmon River	Basic	A-Run
Lemhi River	Intermediate	A-Run
Pahsimeroi River	Intermediate	A-Run
East Fork Salmon River	Intermediate	A-Run
Upper Salmon River	Intermediate	A-Run

Basic application of TRT criteria:

- Six populations must meet viability criteria, one of which must meet high viability criteria
- One of the Large populations (Upper Middle Fork OR Lower Middle Fork Tributaries) must meet viability criteria
- Four additional Intermediate or larger populations must meet viability criteria (all remaining except Secesh and North Fork Salmon River)
- At least one A-run and one B-run must be represented

Considerations:

- South Fork Salmon is the only B-run, intermediate sized population, has no hatchery influence
- Spatial structure should be strongly considered in the choice of populations in this large MPG – those that meet viability criteria should be spread across US, MF and SF and lower Salmon
- A-run populations made up 2/3 of the total populations in this MPG. Where possible, maintaining the distribution of A and B run populations would most closely mirror historical (lower-risk) conditions
- Upper Salmon, EF, Lemhi, Pahsimeroi, Little Salmon/Rapid all have some hatchery influence. This tends to be out of MPG – e.g. Dworshak B, Hells Canyon A.
- Little monitoring on any of these populations except Rapid River
- Secesh, South Fork, Chamberlain and Upper Middle Fork all have no history of hatchery influence, and are relatively natural systems.

TRT Recommendation:

1 Highly Viable and 5 Viable:	Upper Middle Fork Chamberlain South Fork Salmon 2 Additional Intermediate or Large populations 1 Additional population of any size
Maintained:	All remaining extant populations

5. Imnaha MPG

Component populations:

	Size Category	Life History Type
Imnaha River	Intermediate	A-Run

Basic application of TRT criteria:

- One population must meet viability criteria

TRT Recommendation:

Highly Viable:	Imnaha River
Maintained:	N/A

6. Hells Canyon MPG

Component populations:

	Size Category	Life History Type
Hells Canyon	-	-
Powder River (extirpated)	-	-
Burnt River (extirpated)	-	-
Weiser River (extirpated)	-	-

Considerations:

- With the possible exception of several small tributaries in Hells Canyon, this MPG is largely extirpated. Fish that are currently occupying those small tributaries may be the only remnants of this MPG . A key research need is to determine whether these are remnants or hatchery strays. If they are remnants, emphasis should be placed on recovering this population. The other extirpated populations are addressed in the accompanying memo.

C. *Snake River fall chinook salmon*

1. Snake River Mainstem MPG

Component populations:

	Size Category	Life History Type
Lower Mainstem	$Small_{FC}$	-
Marsing Reach (extirpated)	$Large_{FC}$	-
Salmon Falls (extirpated)	$Large_{FC}$	-

Basic application of TRT criteria:

- Two populations must meet viability criteria, both of which must meet high viability criteria

Considerations:

- Two upstream populations are extirpated
- The two upstream populations were historically the most productive
- Additional information about the TRT recommended approach to consideration extirpated areas in recovery planning is presented in the accompanying memo. We recognize that there are significant difficulties in re-establishing fall chinook populations above the Hells Canyon complex, and suggest that initial effort be placed on recovery for the extant population, concurrently with scoping efforts for re-introduction. As recovery efforts progress, the risk and feasibility associated with opening this area to fall chinook can be re-assessed.

TRT Recommendation:

Highly Viable:	Lower Mainstem Marsing Reach OR Salmon Falls
Re-considered as recovery efforts progress:	Marsing Reach or Salmon Falls

D. Snake River sockeye salmon

1. Stanley Lakes Basin

Component populations:

	Size Category	Life History Type
Redfish Lake	-	-
Alturas Lake (extirpated)	-	-
Pettit Lake (extirpated)	-	-
Yellowbelly Lake (extirpated, and of uncertain historical status)	-	-
Stanley Lake (extirpated, and of uncertain historical status)	-	-

Basic application of TRT criteria:

- The IC-TRT required 2/3 of the populations in ESUs with only one MPG to meet viability criteria. This value (2/3) was chosen as a number that was substantially greater than half, with the intent of mitigating for the small number of MPGs with increased numbers of populations. However, there is great uncertainty around the proportion or number of populations that would adequately mitigate risk. With such a small number of populations in this MPG, increasing the number of populations will substantially reduce the risk faced by the ESU. Our next update to our viability criteria will explain the rationale for this recommendation more thoroughly.
-

Considerations:

- Four of five populations are entirely extirpated
- Sockeye are currently maintained in a captive broodstock program, and are at extremely high risk
- Additional information about the TRT recommended approach to extirpated areas will be forthcoming.

TRT Recommendation:

2 Highly Viable and 1 Viable:	Redfish Lake Alturas Lake Pettit Lake
Re-considered as recovery efforts progress:	Yellowbelly Lake Stanley Lake

E. Upper Columbia spring chinook salmon

1. East Cascades MPG

Component populations:

	Size Category	Life History Type
Wenatchee River	Very Large	Spring
Entiat River	Basic	Spring
Methow River	Very Large	Spring
Okanogan River (extirpated)	Basic (U.S. only)	Spring

Basic application of TRT criteria:

- Three populations must meet viability criteria, two of which must meet high viability criteria

Considerations:

- Okanogan River population is extirpated
- Additional information about the TRT recommended approach to extirpated areas will be forthcoming.
- An additional recommendation to moderate risk for an ESU with only one MPG was to require at least 2 populations to meet highly viable status (<1% extinction risk for abundance and productivity). The lowest risk scenario for the ESU would be for the two very large populations (Wenatchee and Methow) to meet highly viable status. Entiat cannot reach these standards due to its inherent spatial structure and the Okanogan population has been extirpated

TRT Recommendation:

2 Highly Viable and 1 Viable:	Wenatchee River (highly viable) Entiat River Methow River (highly viable)
Re-considered as recovery efforts progress:	Okanogan River

F. Upper Columbia steelhead

1. East Cascades MPG

Component populations:

	Size Category	Life History Type
Crab Creek (anadromous component functionally extirpated)	Basic	(Summer A)
Wenatchee River	Intermediate	Summer A
Entiat River	Basic	Summer A
Methow River	Intermediate	Summer A
Okanogan River	Intermediate (Basic for U.S. portion only)	Summer A

Basic application of TRT criteria:

- Three populations must meet viability criteria, two of which must meet high viability criteria
- Two large populations must meet viability criteria

Considerations:

- The anadromous component of Crab Creek was likely historically less robust than those of other populations
- The Okanogan population includes some territory in Canada – for U.S. purposes, this population should meet requirements of a "Basic" population within the U.S., or "intermediate" if status within both countries is considered
- An additional recommendation to moderate risk for an ESU with only one MPG was to require at least 2 populations to meet highly viable status (<1% extinction risk for abundance and productivity). The lowest risk scenario for the ESU would be for the two large populations (Wenatchee and Methow) to meet highly viable status. The Entiat and U.S. Okanogan cannot meet high viability criteria due to their inherent spatial structure, and the anadromous component of Crab Creek has been functionally extirpated.

TRT Recommendation:

2 Highly Viable and 1 Viable:	Wenatchee River Methow River Entiat River Okanogan River
Maintained:	All remaining extant populations
Resident component maintained/reconsidered as recovery efforts progress:	Crab Creek

Mid-Columbia steelhead

1. Cascades Eastern Slopes MPG

Component populations:

	Size Category	Life History Type
White Salmon River (functionally extirpated)	Basic	Unknown
Klickitat River	Intermediate	Summer/Winter
Deschutes River Eastside	Intermediate	Summer
Deschutes River Westside	Large	Summer
Crooked River (extirpated)	Very Large	Summer
Fifteenmile Creek	Basic	Winter
Rock Creek	Basic	Summer

Basic application of TRT criteria:

- Four populations must meet viability criteria, one of which must meet high viability criteria
- Fifteenmile Creek is the only winter population, and thus must meet viability criteria
- One Large or Very Large populations must meet viability criteria. Deschutes River Westside is the only extant population meeting that size requirement.
- In addition, two Intermediate populations must meet viability criteria.

Considerations:

- White Salmon is functionally extirpated. It is blocked by a dam three kilometers upstream, and has been the recipient of abundant hatchery releases from the Skamania stock.

TRT Recommendation:

1 Highly Viable and 3 Viable:	Fifteenmile Creek Deschutes River Westside Klickitat River Deschutes River Eastside
Maintained:	Rock Creek

2. John Day MPG

Component populations:

	Size Category	Life History Type
Lower John Day River	Very Large	Summer
South Fork John Day River	Basic	Summer
Middle Fork John Day River	Intermediate	Summer
North Fork John Day River	Large	Summer
Upper John Day River	Intermediate	Summer

Basic application of TRT criteria:

- Three populations must meet viability criteria, one of which must meet high viability criteria
- Two population in the Large or Very Large size category (Lower John Day and North Fork John Day) must meet viability criteria
- One additional population in the Intermediate (Upper John Day and Middle Fork John Day) category must meet viability criteria

Considerations:

- Lower John Day River population provides an important spatial structure component, as the most downstream population
- North Fork John Day is strong candidate for High Viability status, as it currently appears to be at low risk.
- South Fork John Day is the smallest of the populations

TRT Recommendation:

1 Highly Viable and 2 Viable:	North Fork John Day River Lower John Day River Middle Fork John Day OR Upper John Day
Maintained:	All remaining extant populations

3. Walla Walla-Umatilla MPG

Component populations:

	Size Category	Life History Type
Willow Creek (extirpated)		
Umatilla River	Large	Summer
Walla Walla River	Intermediate	Summer
Touchet River	Intermediate	Summer

Basic application of TRT criteria:

- Two populations must meet viability criteria, one of which must meet high viability criteria
- One Large or Very Large Population (Umatilla River) must meet viability criteria

Considerations:

- Willow Creek population has been extirpated
- Some hatchery influence exists throughout the Walla Walla, Touchet and Umatilla populations.
- Current status suggests that the Walla Walla is closer to meeting viability criteria than the Touchet.

TRT Recommendation:

1 Highly Viable and 1 Viable:	Umatilla River Walla Walla River OR Touchet River
Maintained:	All remaining extant populations

4. Yakima MPG

Component populations:

	Size Category	Life History Type
Satus Creek	Intermediate	Summer
Toppenish Creek	Basic	Summer
Naches River	Large	Summer
Upper Yakima River	Large	Summer

Basic application of TRT criteria:
- Two populations must meet viability criteria, one of which must meet high viability criteria
- One Large or Very Large (Naches or Upper Yakima) population must meet viability criteria

Considerations:
- Having populations at upper and lower ends of the drainage would contribute to a robust spatial structure for the MPG

TRT Recommendation:

1 Highly Viable and 1 Viable:	Naches River OR Upper Yakima
	One of the remaining three populations
Maintained:	All remaining extant populations

Attachment 3:

Examples of Current Status Assessments for Interior Columbia Chinook and Steelhead Populations

Part 1: Wenatchee River Spring Chinook Salmon Population

Part 2: Umatilla River Summer Steelhead Population

Wenatchee River Spring Chinook Salmon Population Current Status Assessment

The Wenatchee Spring Chinook population is part of the Upper Columbia ESU that only has one extant *MPG* including 3 current populations—Wenatchee, Entiat, and Methow Rivers, and one extinct population, the Okanogan (ICTRT 2004). General descriptions of the subbasins and life history characteristics of these populations are provided in the Wenatchee River Subbasin Plan (NPPC, 2004) and the Upper Columbia Recovery Plan (UCSRB 2006).

The Interior Columbia Basin Technical Recovery Team (ICTRT) classified the Wenatchee River Spring Chinook population as "very large" in size based on historical habitat potential (ICTRT 2005). This classification requires a minimum abundance threshold of 2000 wild spawners with sufficient intrinsic productivity (greater than 1.75 r/s measured to spawning) to exceed a 5 % extinction risk on the viability curve (ICTRT 2005). Additionally, the Wenatchee Spring Chinook population was classified as a "type B" population (based on historic intrinsic potential) because it has dendritic tributary structure with multiple major spawning areas (ICTRT 2005).

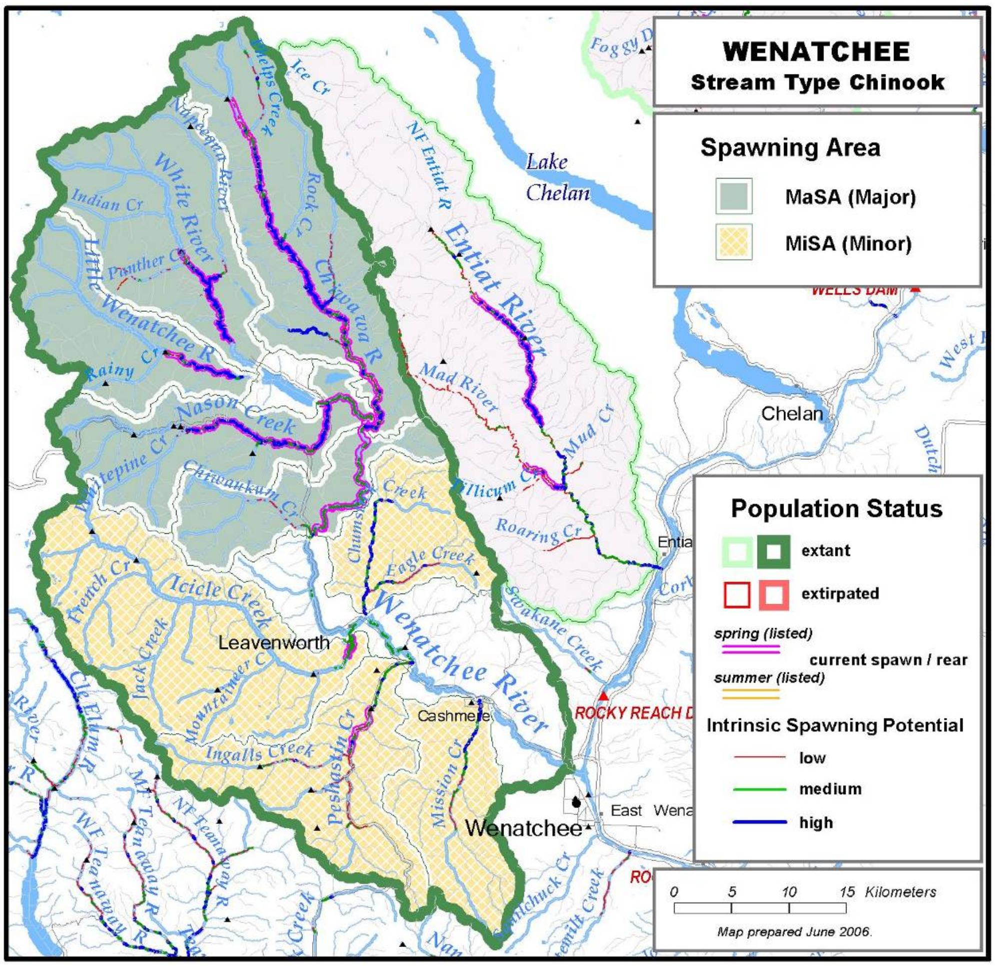

Figure 1. Wenatchee River Spring Chinook Salmon population boundary and major (MaSA) and minor (MiSA) spawning areas.

Table 1. Wenatchee River Spring Chinook Salmon population basin statistics and intrinsic potential analysis summary.

Drainage Area (km2)	3,440
Stream lengths km[a] (total)	1,733.2
Stream lengths km[a] (below natural barriers)	1,082.1
Branched stream area weighted by intrinsic potential (km2)	1.360
Branched stream area km2 (weighted and temp. limited)[b]	1.336
Total stream area weighted by intrinsic potential (km2)	1.883
Total stream area weighted by intrinsic potential (km2) temp limited[b]	1.798
Size / Complexity category	Very Large / B (dendritic structure)
Number of Major Spawning Area	5
Number of Minor Spawning Area	4

[a]All stream segments greater than or equal to 3.8m bankfull width were included

[b]Temperature limited areas were assessed by subtracting area where the mean weekly modeled water temperature was greater than 22^{o}C.

Current Abundance and Productivity

Recent (1960 to 2003) abundance (number of adult spawning in natural production areas) has ranged from 6,718 (1966) to 51 (1995). Abundance estimates are based on expanded redd counts (relatively complete coverage, temporal and spatial components). The results of annual redd surveys are summarized in annual reports and technical memos (e.g., Mosey and Murphy 2002). Prior to 1987, spring chinook redd counts were based on a single survey completed during or after peak spawning activity. The single survey index areas were the most heavily spawned stream reaches. Since 1987, redd counts in the Wenatchee River basin have been based on multiple surveys and include most of the available spawning habitat (Beamesderfer et al., 1997). Since 1995, age composition and hatchery contribution estimates have been based on carcass survey recoveries summarized in the annual WDFW spawning ground survey reports. Prior to 1995 age composition estimates were based on returns to the Leavenworth hatchery facility in Icicle Creek and on samples of sport catch of wild fish (Beamesderfer, et al., 1997). Estimates of the annual number of spawners are derived from the redd count data by applying a standard expansion factor (2.2 fish per redd) based on an average ratio of redd counts above the Chiwawa River weir to direct estimates of the number of spring chinook passing the weir site (Beamesderfer et al., 1997).

Recent year natural spawners include returns originating from naturally spawning parents, strays from the Leavenworth Hatchery program in Icicle Creek and returns from a directed supplementation program (primarily from Chiwawa River releases). The most recent 10 year average contribution of naturally produced returns on the spawning grounds has been 62% (Table 2), ranging from 35% to 92%.

Abundance in recent years has been highly variable; the most recent 12-year geomean number of natural origin spawners was 226. During the period 1960-1999, returns per spawner for spring chinook in the Wenatchee subbasin ranged from 0.06 to 4.59. The most recent 20-year (1979-1998) geometric mean of returns per spawner, adjusted for marine survival and delimited at 75% of the size threshold for this population was 0.74 (Table 2).

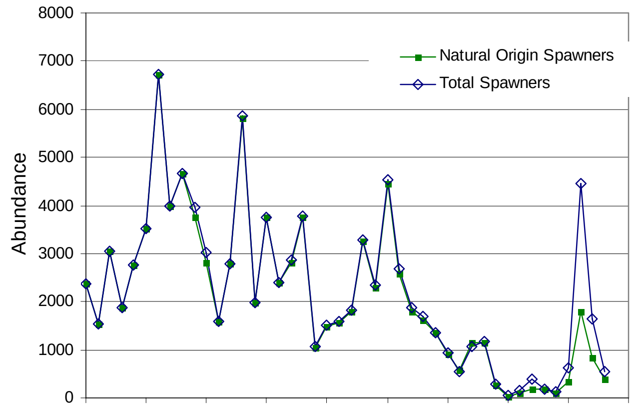

Figure 2. Wenatchee River Spring Chinook Salmon population spawner abundance estimates (1960 to 2003).

Table 2. Wenatchee River Spring Chinook Salmon population abundance and productivity estimates.

10-year geomean natural abundance	226
20-year return/spawner productivity	0.73
20-year return/spawner productivity, SAR adj. and delimited[a]	0.74
20-year Bev-Holt fit productivity, SAR adjusted	1.14
Lambda productivity estimate	1.01
Average proportion natural origin spawners (recent 10 years)	62%
Reproductive success adj. for hatchery origin spawners	No data available

[a]Delimited productivity excludes any spawner/return pair where the spawner number exceeds 75% of the size threshold for this population. This approach attempts to remove density dependence effects that may influence the productivity estimate.

Comparison to Viability Curve

Abundance: 10-year geomean Natural Origin Returns
Productivity: 20-year geomean R/S, SAR adjusted and delimited at 75% of the threshold
Curve: Hockey-Stick curve
Conclusion: Wenatchee Spring Chinook population is at **HIGH RISK** based on current abundance and productivity. The point estimate for abundance and productivity is below the 25% risk curve.

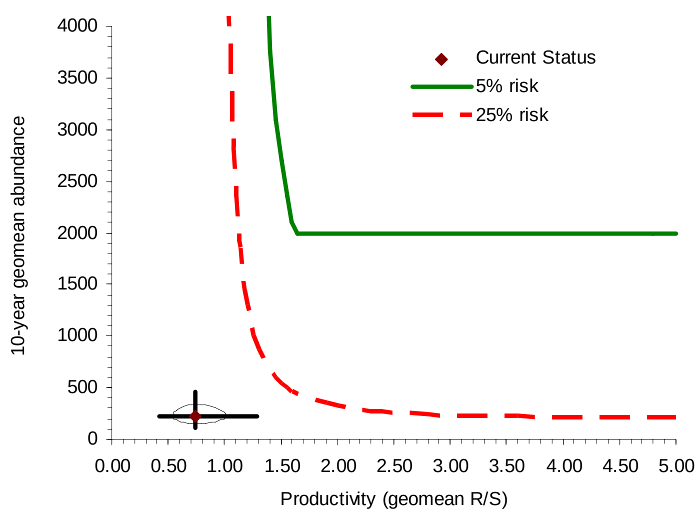

Figure 3. Wenatchee River Spring Chinook Salmon population abundance and productivity compared to the viability curve for this ESU. The point estimate includes a 1 SE ellipse and 95% CI (1.81 X SE abundance line, and 1.80 X SE productivity line).

Spatial Structure and Diversity

The ICTRT has identified five historical major spawning areas (MaSAs) and four minor spawning areas (MiSAs) within the Wenatchee population (Figure 4). The five MaSAs are: Chiwawa, Nason Cr., Little Wenatchee R., White River and the upper Wenatchee mainstem (Tumwater Canyon to Lake Wenatchee). The minor spawning areas (MiSAs) estimated from the intrinsic potential analysis include Icicle, Chumstick, Peshastin, and Mission Creeks.

Currently, the primary spawning areas used by spring Chinook in the Wenatchee are the Chiwawa River, Nason Creek, White River, the Little Wenatchee River and the mainstem Wenatchee between Tumwater Canyon and Lake Wenatchee (Salmonscape 2003; Tonseth 2003). Icicle Creek consistently has unlisted Carson stock spring Chinook spawning below the Leavenworth National Fish Hatchery and, beginning in 2001, Carson stock hatchery spring Chinook have been planted in Peshastin Creek. Redds in these drainages would not contribute to VSP parameters because almost no wild Wenatchee origin fish are known to spawn in these MiSAs. During high abundance years, such as 2001, spring Chinook were also observed in Chiwaukum Creek (A. Murdoch, personal communication).

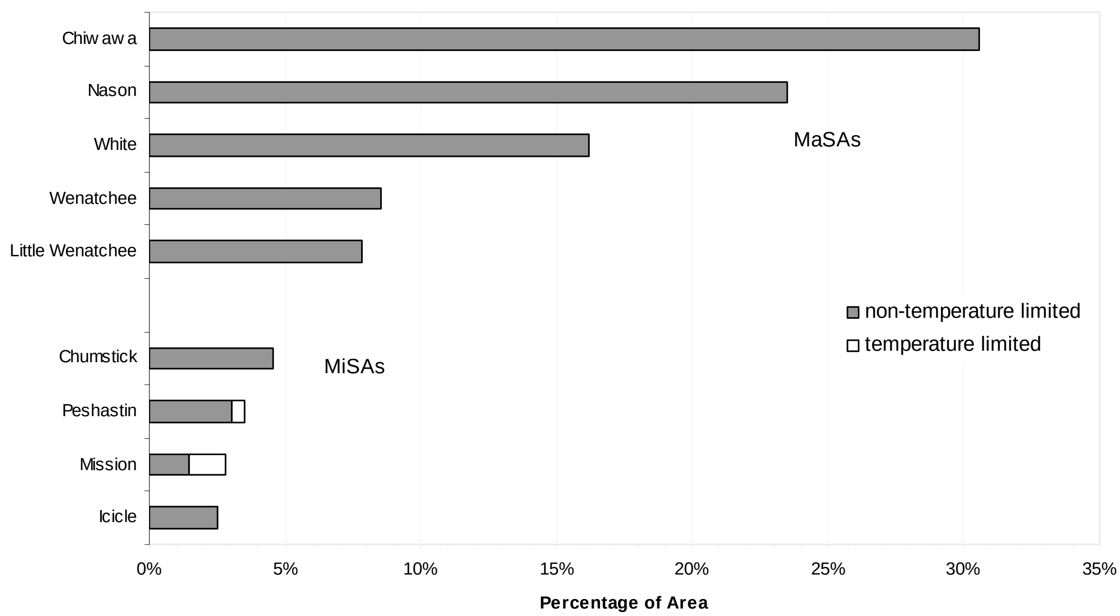

Figure 4. Wenatchee River Spring Chinook Salmon population distribution of intrinsic potential habitat across major and minor spawning areas. White bars represent current temperature limited areas that could potentially have had historical temperature limitations.

Factors and Metrics

A.1.a Number and spatial arrangement of spawning areas

The Wenatchee spring Chinook population has five MaSAs (Chiwawa, Nason, White, and Little Wenatchee, and Upper Wenatchee mainstem) and they are all currently occupied (based on agency defined distribution) so it is at *very low risk*.

A.l.b. Spatial extent or range of population

The Wenatchee spring Chinook population has five MaSAs (Chiwawa, Nason, White, and Little Wenatchee, and Upper Wenatchee mainstem) and they are all occupied (based on agency defined distribution) so it is at *very low risk*. Additionally, based on redd counts in index areas from the most recent brood cycle (2000-2004) and during the last 3 brood cycles, the Wenatchee population would also be at very low risk. However, there were some years during the last 3 brood cycles that did not meet minimum occupancy requirements in the White, Little Wenatchee, and Upper Wenatchee mainstem MaSAs.

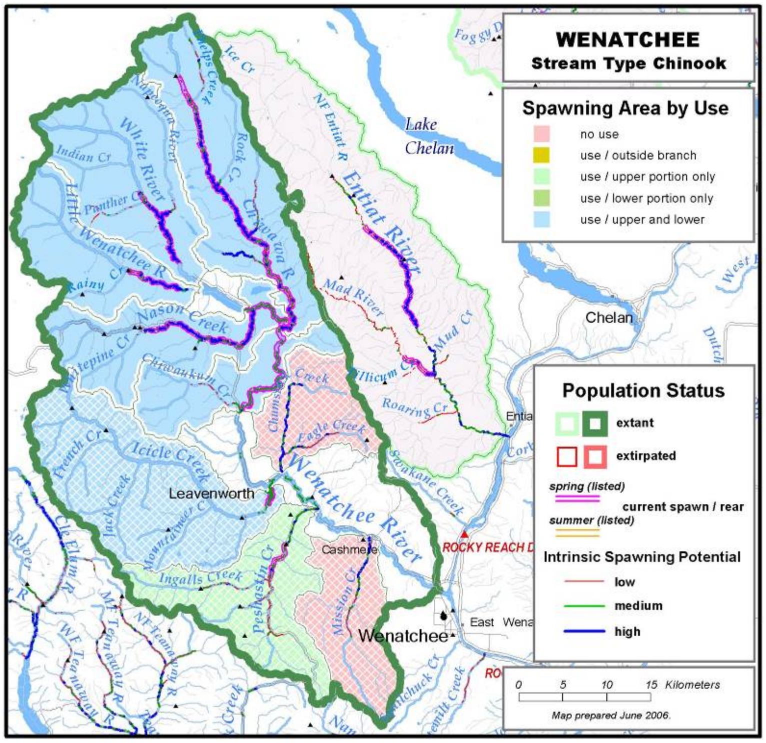

Figure 5. Wenatchee River Spring Chinook Salmon population current spawning distribution and spawning area occupancy designations.

A.1.c. Increase or decrease in gaps or continuities between spawning areas

There has been no increase or decrease in gaps between MaSAs for the Wenatchee spring Chinook population, however the loss of multiple MiSAs at the lower end of the population boundary (below Tumwater Canyon) puts the population at *moderate risk* for this metric. It is assumed that habitat conditions, primarily flow and barriers prohibit the use of Mission and Chumstick Creeks as minor spawning areas. There is considerable uncertainty regarding the ability of these watersheds (Mission and Chumstick) to produce spring Chinook, even under pristine historical conditions. Additionally, there is uncertainty regarding passage of spring Chinook at the Boulder field in Icicle Creek. The opinion of local biologists is that the boulder field always was a barrier (even though road debris has made it artificially enhanced) and recent studies using marked hatchery fish from the LNFH (Cappellini 2001), and historical information from the Wenatchi tribe support that assumption.

B.1.a. Major life history strategies

The Wenatchee spring Chinook population is *very low risk*, because no major life history strategies have been lost.

Studies of juvenile rearing and migration have identified three major juvenile life history patterns within the Wenatchee spring chinook population: summer and overwinter rearing within natal spawning areas, fall presmolt migration and overwintering in the mainstem Wenatchee downstream of natal tributaries, and early summer emigration to downstream areas for summer rearing and overwintering. Limited PIT tagging information indicates that emigrating parr and presmolts use the mainstem reaches above Tumwater Dam for subsequent rearing.

B.1.b. Phenotypic variation

We do not have data available for this metric. Even if we determined that there was a change to one or more traits we do not know what the exact baseline is because changes likely occurred before there was biological monitoring. Therefore, we will assume that there has been some change and increase in variance for 2 or more traits placing the population at *moderate risk*.

B.1.c. Genetic variation

The Wenatchee spring Chinook population was determined to be at *high risk* for genetic variation due to a persistent homogenization from previous fish management efforts. Analyses based on allozymes collected in the 1980s suggest that there was some differentiation between subpopulations consistent with the level of differentiation expected in that time frame, particularly in the White and Twisp drainages. However, microsatellite samples collected in the late 1990s and early 2000s do not show this same differentiation, suggesting that recent management practices and the sequence of extremely low annual spawning numbers in the mid 1990s may have disrupted natural gene flow (ICTRT pop id draft, in prep), A third study (Murdoch et al. date), also analyzed by the ICTRT, includes samples only from the Wenatchee River and indicates that there is some differentiation between watersheds Nason Creek, White River, and Chiwawa River samples. The subgroup concluded that the overall Wenatchee population has been homogenized with other UC populations due to past practices. Their conclusion was based on high similarity to all UC hatchery samples and AMOVA analysis indicating no apparent structure between populations. However, there is some indication, in both the allozyme data and the more recent microsatellite data that there may be some substructure within the population. Data examined include both allozyme and microsatellite data collected by WDFW and analyzed in Ford et al. (2000), and by the ICTRT genetics subgroup (Analyses to be published, available upon request.). It is possible that the true genetic risk metric for this population is lower. If additional data becomes available indicating differentiation between and within populations (either genetic data indicating levels of divergence consistent with the time since separation; or genetic information showing strong spatial structure), the risk level for this metric could improve to moderate or low risk.

B.2.a. Spawner composition

(1) Out-of-ESU spawners. The Wenatchee spring Chinook population is at *high risk* with respect to this metric due to the presence of non-local (outside the ESU origin) stocks on the spawning grounds, which include both LNFH and other stocks from hatcheries outside the Upper Columbia ESU. Tagging studies indicate that LNFH stray rates are generally low (<1%) (Pastor 2004). However, based on expanded carcass recoveries from spawning ground surveys (2001-2004), LNFH and other out-of-basin spawners have comprised from 3-27% of the spawner composition above Tumwater Canyon (WDFW unpublished data). Its possible that 4 years of data is not sufficient to evaluate this metric and our risk assessment could change with the inclusion of a longer time series of data. It has been suggested that the mark rate and recovery rate for hatchery fish was insufficient to determine spawner composition prior to 2000 (Andrew Murdoch, personal communication). Therefore, continuing a 100% external mark rate of hatchery fish and recovering high proportions of carcasses should be a priority.

(2) Out of MPG spawners. The Upper Columbia ESU only has one extant MPG, so this metric is *not applicable* and no score will be given.

(3) *Out of population spawners*. Out of population (but within MPG) origin spawners comprised 0% and 1.8% of the naturally spawning population in 2001 and 2002, respectively (Tonseth 2003, 2004). Based on this short-term data set, the population was at *low risk* with respect to this metric. However, we recognize that two years is likely not sufficient to assess long-term risk and conclude that more years need to be added to the time series. Additionally, if the rearing and release practices discussed in the next metric are not addressed then all the hatchery fish on the spawning grounds will fall into this category and the population will be at high risk for this metric.

(4) Within-population hatchery spawners . Since 1993, a total of 56% of the spawners in tributaries above Tumwater Canyon have been of local hatchery origin, specifically the Chiwawa supplementation program (WDFW unpublished data). Regardless of the duration (# of generations), this high proportion of hatchery fish on the spawning grounds places the population at *high risk* for this metric. Additionally, the Chiwawa River integrated hatchery program strays to other non-target MaSAs and commonly makes up greater than 10 % of the spawner composition in Nason Creek and the White and Little Wenatchee Rivers, based on comprehensive data collected in 2001 and 2002 (Tonseth 2003; Tonseth 2004).

B.3.a. Distribution of population across habitat types.

The intrinsic potential distribution for Wenatchee spring Chinook covered four ecoregions; however, over 90% of the high to medium rated habitat was in two ecoregion types, Chiwaukum Hills and Lowlands and Wenatchee Chelan Highlands. The loss of occupancy in all four MiSAs below Tumwater Canyon did not eliminate an ecoregion type or shift the distribution of ecoregion types by more than 1/3. Therefore, the population was at *low risk* for this metric.

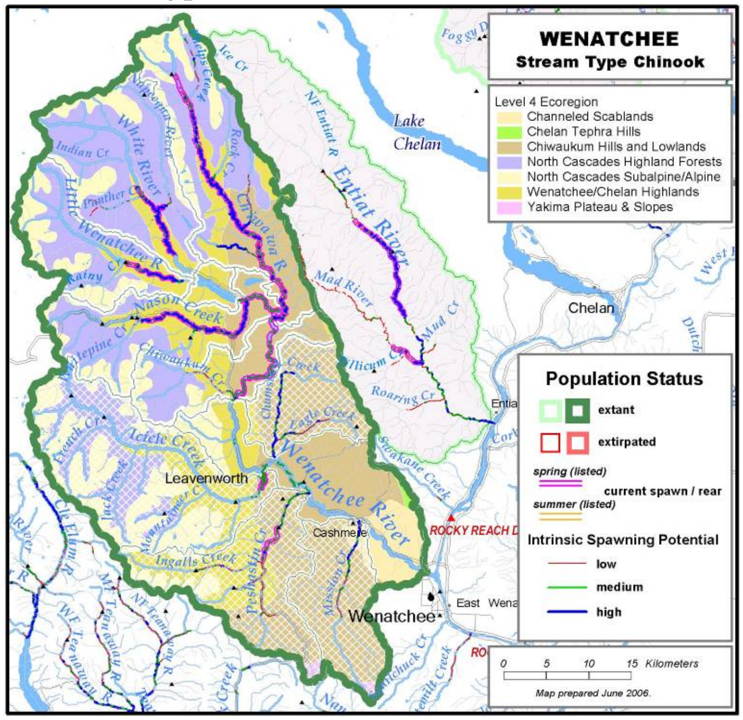

Figure 6. Wenatchee River Spring Chinook Salmon population spawning distribution across EPA level 4 ecoregions.

Table 3. Wenatchee River Spring Chinook Salmon population proportion of current spawning areas across EPA level 4 ecoregions.

Ecoregion	% of historical spawning area in this ecoregion (non-temperature limited)	% of currently occupied spawning area in this ecoregion (non-temperature limited)	% of historical spawning area in this ecoregion (temp. limited)[a]
Channeled Scablands	1.1	0.0	1.1
North Cascades Highland Forests	4.3	3.3	4.3
Wenatchee/Chelan Highlands	41.7	47.6	41.7
Chiwaukum Hills And Lowlands	52.9	49.1	52.9

[a]Temperature limited areas were assessed by subtracting area where the mean weekly modeled water temperature was greater than 22°C.

B.4.a. Selective change in natural processes or selective impacts.

Hydropower system: The hydropower system and associated reservoirs impose some selective mortality on spring Chinook smolt outmigrants, but we assumed negligible effects to upstream migrating adults. Current estimates of project mortality are approximately 2%, but some portion of that 2% is natural mortality, and we assumed that the mortality was not selective against either early or late returning adults. For the smolt effects we assumed that hydro project mortality, reservoir delays, and size selective predation imposed selective mortality against smaller smolts (Baldwin et al. 2003; Fritz and Pearsons 2006). The specific magnitude of selective mortality and the exact proportion of population that is affected are unknown. The duration of the impact was considered long because it is ongoing and has been occurring for multiple generations. We rated the selection intensity as high because the proportion of the population effected was high due to cumulative smolt mortality in the hydrosystem. We rated the heritability as low because smolt size is primarily a function of environmental conditions. The resulting selectivity rating for the hydrosystem was **moderate risk**.

Harvest: Mainstem fishery harvest rates on returning Upper Columbia spring chinook (including the Wenatchee run) have ranged from 3.5% to 14.8% during the period 1980 to 2005, averaging approximately 10% annually (ODFW & WDFW, 2006). Although some harvest may be size selective for larger fish, the selective mortality is assumed to affect less than 2% of the population resulting in a rating of negligible for the proportion affected. There is no in-basin harvest of Wenatchee spring Chinook. Therefore, the harvest selectivity rating was **low risk**.

Hatcheries: The Chiwawa River hatchery program is operated to be non-selective by collecting broodstock so that their run-timing, sex, and age mimic that of the total run at Tumwater Dam (Wenatchee HGMP). This metric was rated at **low risk**.

Habitat: There are two habitat changes that we considered for selective mortality, altered flow profiles and decreased rearing habitat in the lower Wenatchee River mainstem. The timing of altered flow profiles is such that it does not affect run timing for returning adults so it was rated at **low risk**. We also considered the loss of diversity of juvenile life history pathways due to the loss of side channels, riparian condition, and floodplain function in the lower Wenatchee mainstem. A relatively high proportion of subyearling spring Chinook are known to migrate from the tributaries (Chiwawa) in the fall and overwinter in the Upper Wenatchee mainstem and Tumwater Canyon (e.g., Murdoch et al., 1999). It is uncertain weather or not the Lower Wenatchee River downstream of Tumwater Canyon was a historically important winter rearing area. If it was then the selectivity rating for this metric would be moderate or high risk. However, given the uncertainty of the historic utilization of the Lower Wenatchee River we rate this metric at **low risk**.

The overall selectivity rating is **moderate risk**.

Spatial Structure and Diversity Summary

The Wenatchee spring Chinook population was determined to be at low risk for goal A (allowing natural rates and levels of spatially mediated processes) but at high risk for goal B (Maintaining natural levels of variation) resulting in an overall HIGH risk rating. The metrics for genotypic and phenotypic variation were the determining factors for the high risk rating of Wenatchee spring Chinook. We concluded that there was evidence for a high degree of homogenization within the Wenatchee population as well as among the three extant Upper Columbia Spring chinook populations. However, there was considerable uncertainty regarding whether or not the level of divergence in the Wenatchee was sufficient for a moderate risk rating. Therefore continued efforts to maintain natural levels of exchange within and among populations and further evaluation could lead to an improved risk rating. For B.1.b. (phenotypic variation), an analysis needs to be conducted that shows that the phenotypic traits of the current population are consistent with the assumed historical condition or with unaltered reference populations in a similar habitat, geologic, and hydrologic setting. Based on the scoring system, this metric must be addressed in order for the status of goal B to improve to low risk.

There were two metrics that were rated at high risk related to spawner composition that did not directly reduce the overall risk conclusion, but should be considered potential threats to both genotypic (B.1.3) and phenotypic variation (B.1.b). First, Chiwawa River hatchery fish (local origin stock; B.2.a.2) comprise a large portion of the fish on the spawning grounds over multiple generations. Additionally, this hatchery operation is not meeting best management practices because the rearing and release strategies (acclimation of Chiwawa fish on Wenatchee River water over the winter) increase the probability of straying to non-target MaSAs. Second, the high proportion (3-27 %) of LNFH fish (out-of-ESU stock) on the spawning grounds poses an additional risk to genotypic and phenotypic variation. However, due to the scoring system these high-risk ratings were averaged with other metrics and did not directly cause an increased risk rating.

Table 4. Wenatchee River Spring Chinook Salmon population spatial structure and diversity risk rating summary.

<table>
<tr><th rowspan="2">Metric</th><th colspan="5">Risk Assessment Scores</th></tr>
<tr><th>Metric</th><th>Factor</th><th>Mechanism</th><th>Goal</th><th>Population</th></tr>
<tr><td>A.1.a</td><td>VL (2)</td><td>VL (2)</td><td rowspan="3">Mean = 1.33 Low Risk</td><td rowspan="3">Low Risk</td><td rowspan="12">High Risk</td></tr>
<tr><td>A.1.b</td><td>VL (2)</td><td>VL (2)</td></tr>
<tr><td>A.1.c</td><td>M (0)</td><td>M (0)</td></tr>
<tr><td>B.1.a</td><td>VL (2)</td><td>VL (2)</td><td rowspan="3">High Risk (-1)</td><td rowspan="9">High Risk</td></tr>
<tr><td>B.1.b</td><td>M (0)</td><td>M (0)</td></tr>
<tr><td>B.1.c</td><td>H (-1)</td><td>H (-1)</td></tr>
<tr><td>B.2.a(1)</td><td>H (-1)</td><td rowspan="4">H
(-1)</td><td rowspan="4">High Risk (-1)</td></tr>
<tr><td>B.2.a(2)</td><td>NA</td></tr>
<tr><td>B.2.a(3)</td><td>L (1)</td></tr>
<tr><td>B.2.a(4)</td><td>H (-1)</td></tr>
<tr><td>B.3.a</td><td>L (1)</td><td>L (1)</td><td>Low Risk (1)</td></tr>
<tr><td>B.4.a</td><td>M (0)</td><td>M (0)</td><td>Moderate Risk (0)</td></tr>
</table>

<u>Overall Risk Rating:</u>

The Wenatchee spring Chinook population is not currently meeting viability criteria. Of particular concern is the high risk rating with respect to abundance and productivity. The population cannot achieve any level of viability without improving its status on the viability curve for both abundance and productivity. Spatial structure and diversity was also rated at high risk, due primarily to a high level of genetic homogenization within and among populations. Improvement of the spatial structure and diversity status to low risk would be required to allow the Wenatchee population to achieve a "highly viable" status (in addition to the improvements needed for abundance and productivity). Based on the MPG guidelines, the Wenatchee population will need to achieve a highly viable status for recovery of the ESU (ICTRT 2005).

		Spatial Structure/Diversity Risk			
		Very Low	**Low**	**Moderate**	**High**
Abundance/ Productivity Risk	**Very Low** (<1%)	**HV**	**HV**	**V**	M*
	Low (1-5%)	**V**	**V**	**V**	M*
	Moderate (6 – 25%)	M*	M*	M*	
	High (>25%)				Wenatchee

Figure 7. Wenatchee River Spring Chinook Salmon risk ratings integrated across the four viable salmonid population (VSP) metrics. *Viability Key: HV – Highly Viable; V – Viable; M* – Candidate for Maintained; Shaded cells – does not meet viability criteria (darkest cells are at highest risk).*

Wenatchee River Spring Chinook Salmon – Data Summary

Data type: Redd count expansions (Wenatchee Spring Chinook without Icicle Creek). Natural returns include wild origin spawners removed as broodstock for short-term supplementation actions.

SAR: Expanded Chiwawa SAR index

Table 5. Wenatchee River Spring Chinook Salmon population abundance and productivity data used for curve fits and R/S analysis. Bolded values were used in estimating the current productivity (Table 6).

Brood Year	Spawners	%Wild	Natural Run	Nat. Rtns	R/S	SAR Adj. Factor	Adj. Rtns	adj R/S
1979	**1063**	**0.98**	**1039**	**1406**	**1.32**	**1.32**	**1859**	**1.75**
1980	1519	0.98	1486	3025	1.99	0.80	2408	1.58
1981	1595	0.98	1566	4045	2.54	0.74	2977	1.87
1982	1819	0.98	1786	2873	1.58	0.72	2062	1.13
1983	3286	0.99	3249	1693	0.52	0.80	1358	0.41
1984	2341	0.98	2295	1105	0.47	1.36	1506	0.64
1985	4529	0.98	4445	1380	0.30	1.34	1846	0.41
1986	2674	0.97	2582	886	0.33	1.80	1597	0.60
1987	1878	0.96	1803	1065	0.57	1.48	1575	0.84
1988	1692	0.96	1625	696	0.41	0.73	505	0.30
1989	**1349**	**0.96**	**1347**	**829**	**0.61**	**1.27**	**1054**	**0.78**
1990	**927**	**0.95**	**899**	**183**	**0.20**	**3.12**	**572**	**0.62**
1991	**552**	**1.00**	**582**	**122**	**0.22**	**7.30**	**890**	**1.61**
1992	**1080**	**0.98**	**1140**	**70**	**0.06**	**5.21**	**364**	**0.34**
1993	**1179**	**0.89**	**1146**	**124**	**0.11**	**0.49**	**61**	**0.05**
1994	**275**	**0.89**	**255**	**205**	**0.75**	**1.92**	**394**	**1.43**
1995	**51**	**0.35**	**18**	**229**	**4.53**	**0.41**	**95**	**1.88**
1996	**158**	**0.64**	**109**	**506**	**3.20**	**0.37**	**189**	**1.19**
1997	**385**	**0.40**	**188**	**1768**	**4.59**	**0.15**	**264**	**0.69**
1998	**183**	**0.88**	**174**	**686**	**3.76**	**0.19**	**132**	**0.72**
1999	119	0.92	109					1.75
2000	620	0.55	351					
2001	4446	0.38	1798					
2002	1651	0.51	842					
2003	539	0.71	383					

Table 6. Wenatchee River Spring Chinook Salmon population geometric mean abundance and productivity estimates (values used for current productivity and abundance are shown in boxes).

	R/S measures				Lambda measures		Abundance
	Not adjusted		SAR adjusted		Not adjusted		Nat. origin
delimited	median	75% threshold	median	75% threshold	1987-1998	1979-1998	geomean
Point Est.	0.77	0.75	0.74	0.74	1.02	1.01	226
Std. Err.	0.52	0.47	0.34	0.31	0.65	0.40	0.40
count	10	11	10	11	12	20	10

Table 7. Wenatchee River Spring Chinook Salmon population stock-recruitment curve fit parameter estimates. Biologically unrealistic or highly uncertain values are highlighted in grey.

	Not adjusted for SAR							Adjusted for SAR						
SR Model	a	SE	b	SE	adj. var	auto	AICc	a	SE	b	SE	adj. var	auto	AICc
Rand-Walk	0.73	0.20	n/a	n/a	0.60	0.77	69.6	0.73	0.14	n/a	n/a	0.67	0.16	54.1
Const. Rec	675	173	n/a	n/a	n/a	n/a	66.9	675	171	n/a	n/a	n/a	n/a	66.3
Bev-Holt	3.49	3.58	1001	449	0.38	0.82	67.4	1.14	0.44	2650	1929	0.59	0.23	54.7
Hock-Stk	2.52	1.39	314	193	0.42	0.82	68.9	0.73	0.13	8959	0	0.67	0.16	56.8
Ricker	1.30	0.54	0.00040	0.00023	0.50	0.79	69.5	1.02	0.29	0.00023	0.00016	0.60	0.21	54.9

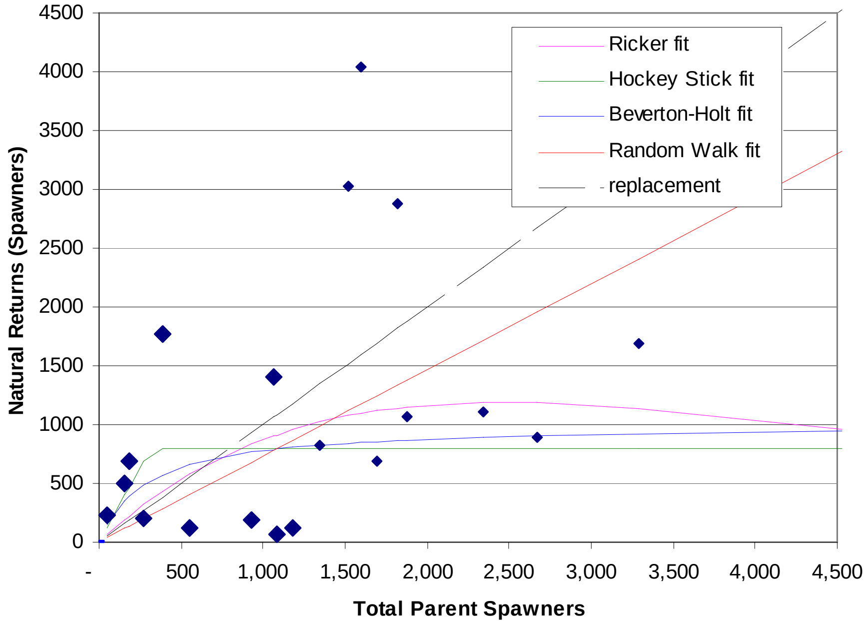

Figure 8. Wenatchee River Spring Chinook Salmon population stock recruitment curves. Bold points were used in estimating the current productivity. Data were not adjusted for marine survival.

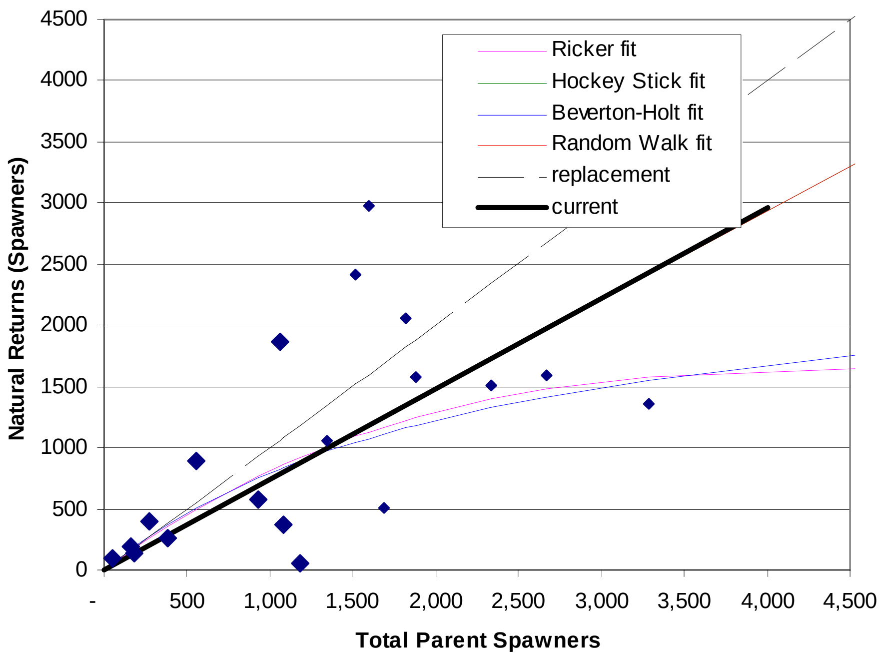

Figure 9. Wenatchee River Spring Chinook Salmon population stock recruitment curves. Bold points were used in estimating the current productivity. Data were adjusted for marine survival.

Literature Cited

Baldwin, C. M., J. G. McLellan, M. C. Polacek, and K. Underwood. 2003. Walleye predation on hatchery releases of kokanees and rainbow trout in Lake Roosevelt, Washington. North American Journal of Fisheries Management 23: 660-676.

Beamesderfer, R. C. P., H. P. Schaller, M. P. Zimmerman, C. E. Petrosky, O. P. Langness, and L. LaVoy. 1997. Spawner-recruit data for spring and summer Chinook salmon populations in Idaho, Oregon, and Washington. Unpublished draft manuscript dated 23 June 1997, 78 p

Cappellini, M. J. 2001. Movements of bull trout (*Salvelinus confluentus*), spring chinook (*Oncorhynchus tshawytscha*), and steelhead (*Oncorhynchus mykiss*) in Icicle Creek, Washington. United States Fish and Wildlife Service, Leavenworth, Washington.

Fritz, A. L. and T. N. Pearsons. 2006. Effects of predation by nonnative smallmouth bass on native salmonid prey: the role of predator and prey size. Transactions of the American Fisheries Society 135:853-860.

ICTRT, 2003. Independent Populations of chinook, steelhead and sockeye for listed Interior Columbia Basin ESUs. Draft. Interior Columbia Basin Technical Recovery Team Report. July, 2003

ICTRT. 2005. Viability Criteria for application to interior Columbia basin ESUs. Interior Columbia Basin Technical Recovery Teamdraft Report. 49 pp.

Mosey, T.R. and L.J. Murphy. 2002. Spring and summer chinook spawning ground surveys on the Wenatchee River basin, 2001. Chelan County Public Utility Dist. 35 p + appendices

Murdoch, A, K. Petersen, T. Miller, M. Tonseth and T. Randolph. 1999. Freshwater production and emigration of juvenile spring chinook salmon from the Chiwawa River in 1998. Washington Dept. of Fish and Wildlife Report. Prepared for Public Utility Dist. #1 of Chelan County. Rept. # SS99-05. 57 p.

NPPC 2004. Wenatchee Subbasin Plan. http://www.nwcouncil.org/fw/subbasinplanning/wenatchee/plan. Portland, Oregon.

Pastor, S. M. 2004. An evaluation of fresh water recoveries of fish released from National Fish Hatcheries in the Columbia River Basin, and observations of straying. American Fisheries Society Symposium XX:xxx-xxx. Bethesda, Maryland.

Oregon Dept. of Fish and Wildlife, Washington Dept. of Fish and Wildlife. 2006. Joint staff report concerning commercial seasons fro spring chinook, steelhead, sturgeon, shad, smelt, and other species and miscellaneous regulations for 2006. Jan. 18,2006.

Salmonscape. 2003. Salmonid Fish Distribution - salmonscape.fishdist. Available from Washington Department of Fish and Wildlife. http://wdfw.wa.gov/mapping/salmonscape/ Olympia, Washington.

Tonseth, M. 2003. 2001 Upper Columbia River stock summary for sockeye, spring Chinook, and summer Chinook. A technical memo to “distribution list”, from the Washington Department of Fish and Wildlife dated January 15, 2003. Wenatchee, Washington

Tonseth, M. 2004. 2002. Upper Columbia River stock summary for sockeye, spring Chinook, and summer Chinook. A technical memo to “distribution list”, from the Washington Department of Fish and Wildlife dated August 31, 2004. Wenatchee, Washington.

UCSRB. 2006. Draft Upper Columbia Spring Chinook Salmon, Steelhead, and Bull Trout Recovery Plan. Prepared for the Washington State Governors Salmon Recovery Office by the Upper Columbia Salmon Recovery Board. Available online: http://okanogancounty.org/water/salmon%20recovery;%20draft%20review%20corner.htm

Umatilla River Summer Steelhead Population Current Status Assessment

The Umatilla River Summer Steelhead population (Figure 1) is part of the Mid-Columbia Steelhead DPS which has four major population groupings (MPG): Cascades Eastern Slope Tributaries, John Day River, Umatilla/Walla Walla Rivers, and the Yakima River group. There are three life history categories in the DPS: summer run, winter run, and summer-winter run combination. The Umatilla River population is a summer run and resides in the Umatilla/Walla Walla Rivers MPG along with the Walla Walla River and Touchet River populations.

The ICTRT classified the Umatilla River population as a "large" sized population (Table 1). A steelhead population classified as large has a mean minimum abundance threshold of 1,500 with sufficient intrinsic productivity (greater than 1.26 recruits per spawner at the minimum abundance threshold) to achieve a 5% or less risk of extinction over a 100-year timeframe.

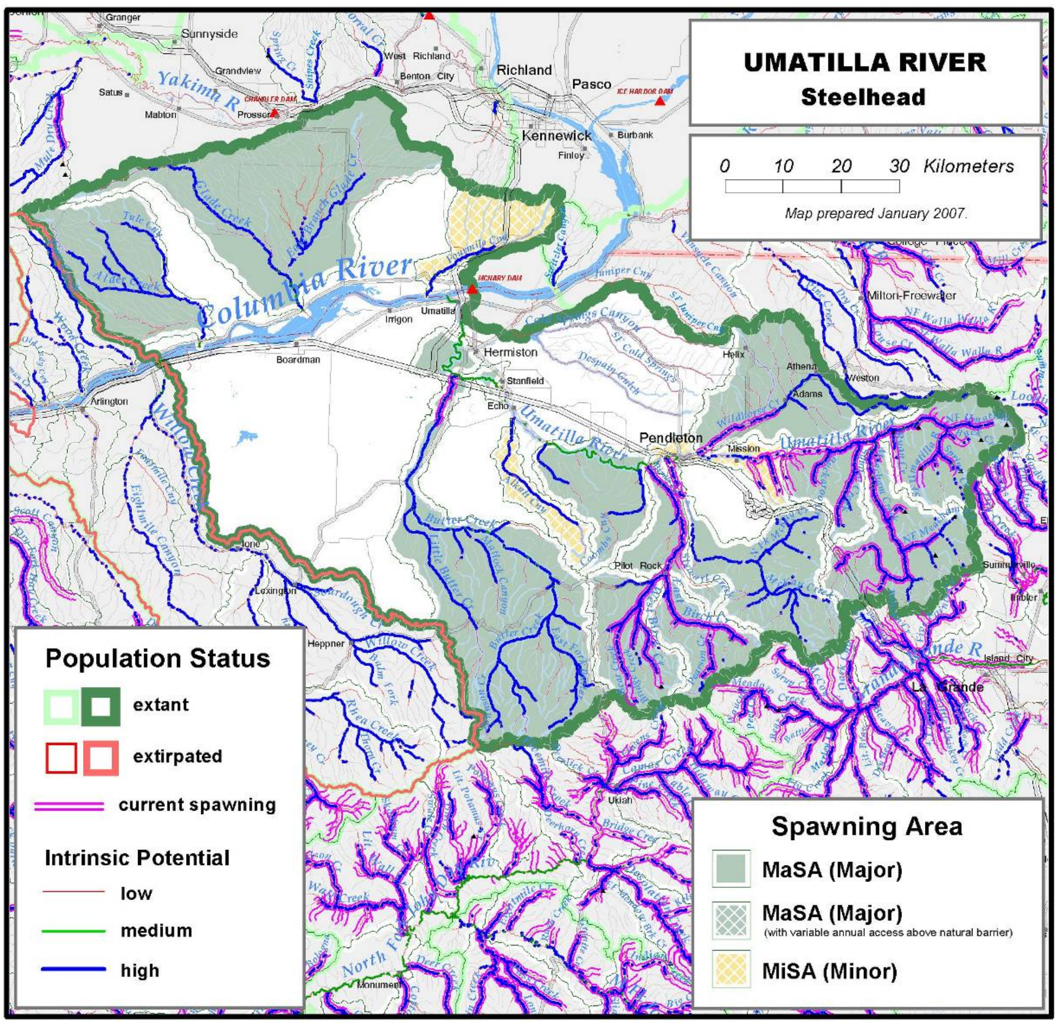

Figure1. Umatilla River Summer Steelhead population boundary and major (MaSA) and minor (MiSA) spawning areas.

Table 1. Umatilla River Summer Steelhead basin statistics and intrinsic potential analysis summary.

Drainage Area (km2)	10,457
Stream lengths km[a] (total)	2,322
Stream lengths km[a] (below natural barriers)	2,278
Branched stream area weighted by intrinsic potential (km2)	7.531
Branched stream area km2 (weighted and temp. limited[b])	7.456
Total stream area weighted by intrinsic potential (km2)	9.070
Total stream area weighted by intrinsic potential (km2) temp limited[b]	3.415
Size / Complexity category	Large / B (dendritic structure)
Number of Major Spawning Areas	13
Number of Minor Spawning Areas	3

[a]All stream segments greater than or equal to 3.8m bankfull width were included

[b]Temperature limited areas were assessed by subtracting area where the mean weekly modeled water temperature was greater than 22°C.

Current Abundance and Productivity

Current (1967 to 2004) total abundance (number of adult spawners in natural production areas) has ranged from 771 (1998) to 5,172 (2002) (Figure 2). Spawner abundance estimates for natural and hatchery summer steelhead in the entire Umatilla River Basin were determined from complete counts of adult returns to Three Mile Falls Dam (TMFD) at river mile 3.7 minus removals or mortality at and above the dam in all years except brood years (BY) 1984-1987. Fish were enumerated using electronic counters from BY 1967-1983, trapping from BY 1988-2000, and a combination of trapping and video monitoring from BY 2001-present. For BYs 1984-1987 abundance estimates were made with mark-recapture estimates. Missing abundance data for BY 1971, 1972, and 1979 were reconstructed using the known mean brood age structure from BY 1991-1998 and all available counts of brood returns in years before and after the missing counts. Counts in BY 1976 and 1978 were also incomplete but not reconstructed. In these years, electronic counters only operated from Dec 24 – May 31 and Dec 13 – Mar 9, respectively. Age structure was determined by reading about 100-150 scales per year collected from adults returning in BY 1994-2004. Missing run year age structure data before BY 1994 was estimated as the BY 1994-2004 mean age structure.

Several sets of missing data for removals and mortalities at and above TMFD were estimated from the best available data. Missing harvest removals were estimated from creel survey data collected from the non-tribal fishery from BY 1993-2004 and the tribal fishery from BY 1993-2001. Harvest of hatchery fish from BY 1988-1992 was estimated as the mean percent harvest of the hatchery run passed above TMFD from the later time period (2.5% non-tribal and 6.4% tribal). All harvested fish were assumed to be natural origin before BY 1988. For years when harvest of natural fish was allowed in the non-tribal fishery (before BY 93), harvest was estimated as mean percent catch of the natural run passed above TMFD (6.8 %) (1993-2004) corrected by the mean percent of catch released (26%). Tribal harvest for BYs 1967-1987 of hatchery and natural steelhead was estimated as their respective mean percent harvest of their runs passed above TMFD (6.7% of the combined natural and hatchery run passed above TMFD). Missing broodstock removals in BY 1981 and 1982 were estimated as one natural fish collected for brood per 750 smolts produced based on the ratio of brood collected and smolts released in the early 1980's. All 95 hatchery fish collected for brood in BY 1991 were assumed to be coded-wire tagged and included in the total removal of 124 hatchery fish at TMFD for coded-wire tag recovery.

Recent year natural spawners include returns originating from naturally spawning parents, Umatilla River hatchery origin fish and out-of-DPS spawners, primarily from the Snake River Basin. Natural origin fish have comprised an average of 73% of natural spawners since hatchery returns have been documented in 1988. Since that time, the percentage of natural origin spawners has ranged from 41% to 96%.

Abundance in recent years has been moderately variable, the most recent 10-year geomean number of natural origin spawners was 1,472 (2,347 total spawners). During the period 1967-2000, returns per spawner for steelhead in the Umatilla River ranged from 0.3 (1978) to 4.98 (1998). The most recent 20-year (1981-2000) geometric mean of returns per spawner SAR adjusted and delimited at 75% of the threshold was 1.50 (Table 2).

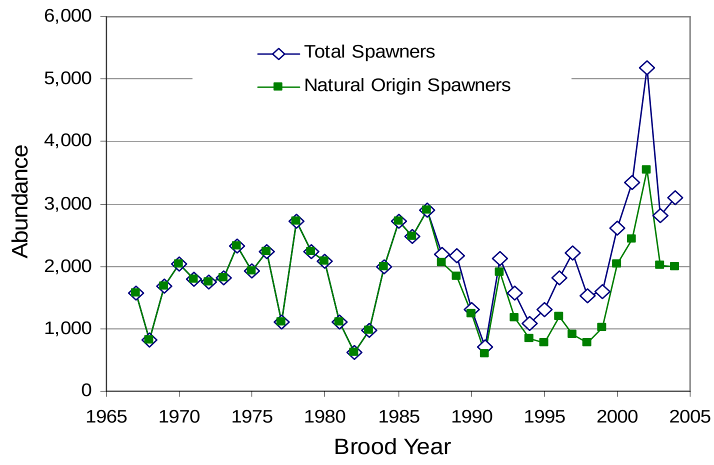

Figure 2. Umatilla River Summer Steelhead population spawner abundance estimates (1967-2004).

Table 2. Umatilla River Summer Steelhead population abundance and productivity measures.

10-year geomean natural abundance	1472
20-year return/spawner productivity	0.94
20-year return/spawner productivity, SAR adj. and delimited[a]	1.50
20-year Bev-Holt fit productivity, SAR adjusted	n/a
Lambda productivity estimate	1.06
Average proportion natural origin spawners (recent 10 years)	0.73
Reproductive success adj. for hatchery origin spawners	n/a

[a]Delimited productivity excludes any spawner/return pair where the spawner number exceeds 75% of the threshold. This approach attempts to remove density dependence effects that may influence the productivity estimate.

Comparison to the Viability Curve

- Abundance: 10-year geomean Natural Origin Returns
- Productivity: 20-yr geomean R/S (adjusted for marine survival and delimited at 1,125 spawners)
- Curve: Hockey-Stick curve
- Conclusion: Umatilla Summer Steelhead population is at MODERATE RISK. The productivity is at low risk because the point estimate is above 5% risk level and the adjusted standard error is above the 25% risk level. Abundance is moderate because the point estimate is slightly below the 5% risk level (Figure 3).

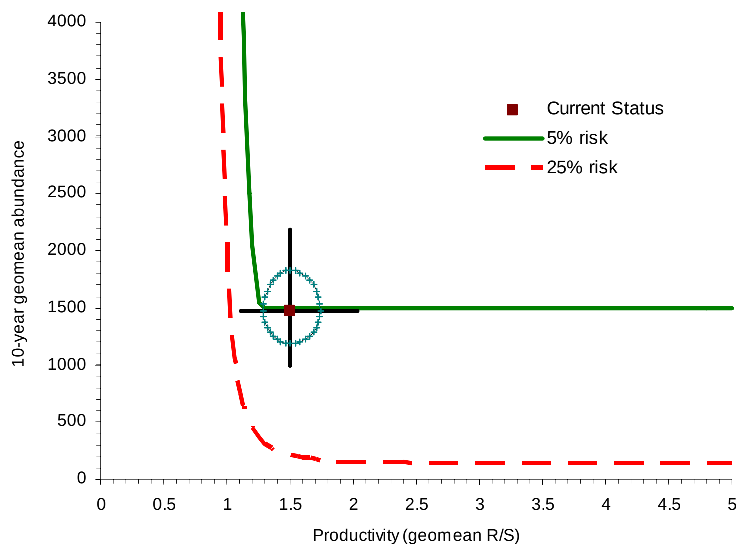

Figure 3. Umatilla River Summer Steelhead current estimate of abundance and productivity compared to the viability curve for this ESU. The point estimate includes a 1 SE ellipse and 95% CI (1.81 X SE abundance line, and 2.02 productivity line).

Spatial Structure and Diversity

The ICTRT has identified 13 historic major spawning areas (MaSAs) and three minor spawning areas (MiSAs) within the Umatilla River steelhead population. In addition, two MaSAs (Alder Creek and Glade Creek) and one MiSA (Fourmile Canyon) were included in the Umatilla River population that are direct tributaries to the Columbia River on the Washington side of the Columbia. We do consider these areas in the assessment of spatial structure/diversity for the Umatilla steelhead population (Figure 4). Current spawning distribution is somewhat limited relative to historic and is concentrated in Birch Creek, Iskulpa Creek, Meacham Creek, Upper Umatilla River, and the North and South Forks of the Umatilla River. There is documented recent year spawning in both Glade Creek and Alder Creek subbasins (Yakama Indian Nation Fisheries Program, 2005).

Spawners within the Umatilla River population include natural-origin returns, hatchery returns of Umatilla River origin broodstock, and hatchery strays, primarily originating from the Snake River Basin. Hatchery-origin fish comprise a significant proportion of the natural spawning fish in most recent years.

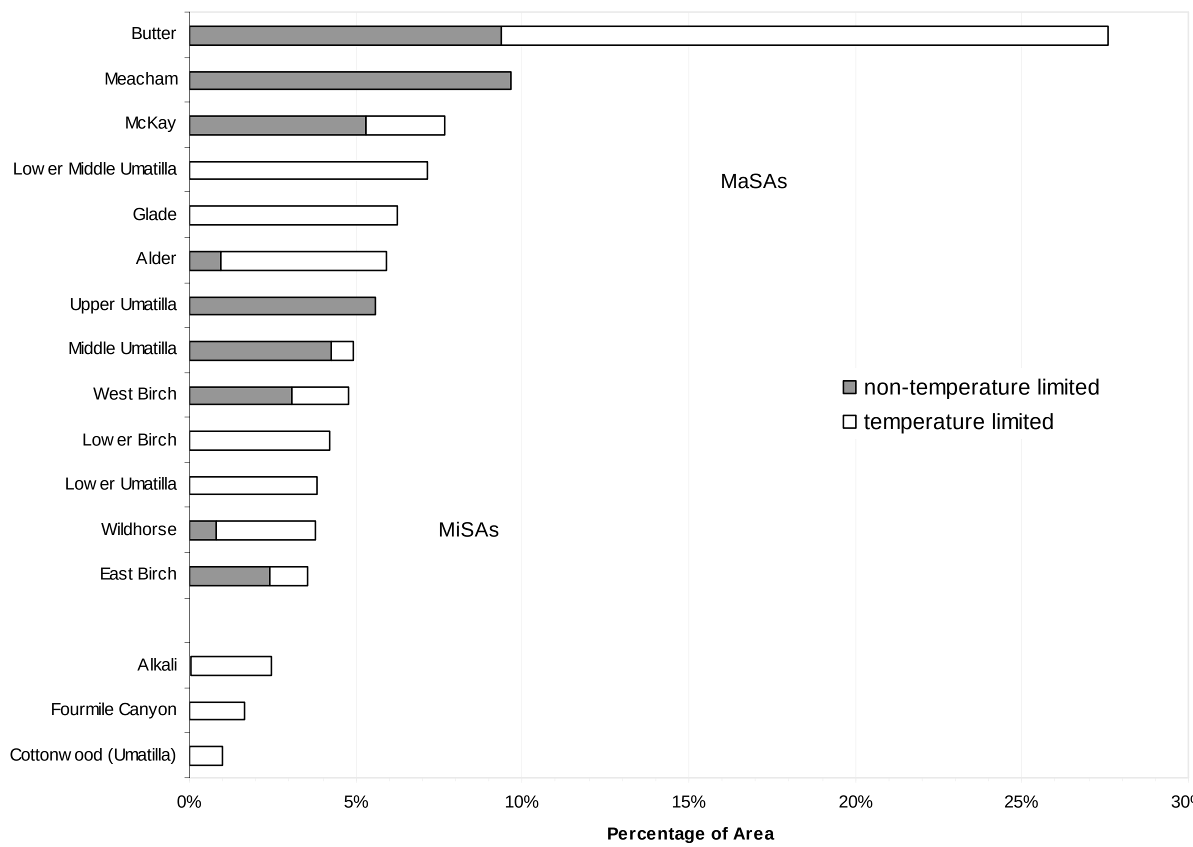

Figure 4. Umatilla River Summer Steelhead distribution of intrinsic potential habitat across major and minor spawning areas. White bars represent current temperature limited areas that could potentially have had historical temperature limitations.

Factors and Metrics

A.1.a. Number and spatial arrangement of spawning areas.

The Umatilla River population has 13 MaSAs and three MiSAs which are distributed in a complex dendritic pattern. Historically the major production areas included Butter Creek, Meacham Creek, McKay Creek, Iskulpa Creek, Birch Creek, and the middle and upper Umatilla River. Spawning distribution has been reduced significantly from the intrinsic historic distribution. Currently eight of the 13 MaSAs are occupied. Alder Creek, Glade Creek, Lower Umatilla, Lower Middle Umatilla, and McKay MaSAs are unoccupied. One of the three MiSAs is currently occupied (Cottonwood Creek). Although there has been a significant reduction in spawner distribution, the Umatilla population rates at **very low risk** because it has more than four occupied MaSAs in a dendritic configuration.

A.l.b. Spatial extent or range of population.

The current spawner distribution is reduced substantially from the intrinsic distribution. Based on the ODFW spawner database and WDFW information, eight of 13 (61.5%) MaSAs are currently occupied and only one of the three MiSAs is occupied (Figure 5). The spatial extent and range of spawning distribution has been reduced to an extent that this population rates as **moderate risk** for this metric. There are 12 index area spawning survey sites in the Umatilla population. Recent survey results will be analyzed for use in future viability assessments.

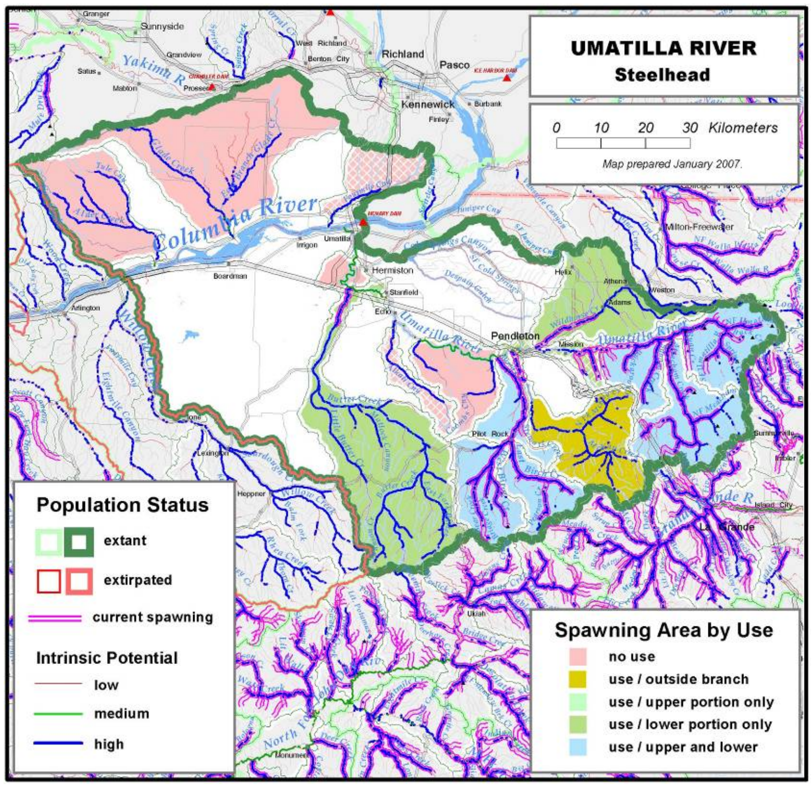

Figure 5. Umatilla River Summer Steelhead population current spawning distribution and spawning area occupancy designations.

A.1.c. Increase or decrease in gaps or continuities between spawning aggregates.

There has been a change in gaps and continuity as a result of the loss of spawning in the McKay Creek and Lower Middle Umatilla River drainages as well as very limited production in the lower portion of the Butter Creek MaSA. Although some spawning occurs in lower Butter Creek, habitat conditions are such that no significant sustained production occurs. Due to the low level of production in Butter Creek it does not serve any connectivity role within or between populations. In addition, less that 75% of the intrinsic MaSAs are currently occupied, thus the rating is **moderate risk** for this metric.

B.1.a. Major life history strategies

We have no observational data to allow any direct comparisons of historic and current life history strategies. Therefore we have used EDT analyses and habitat conditions to infer loss of life history strategies. Flow and temperature changes in the Umatilla Basin have limited movement patterns for both juvenile and adult steelhead. Juvenile steelhead cannot move into some mainstem rearing reaches above McKay Creek for over summer rearing due to high temperatures. Adults are unable to enter the Umatilla in early fall in many years because of the lack of flow as well as high water temperatures. Large areas, such as Butter and McKay creeks drainages, no longer support production. Flow enhancement projects have improved conditions for adult fall migration and summer rearing, particularly below McKay Creek. Past habitat changes have undoubtedly reduced diversity in life history pathways. However, it does not appear that any major pathways have been lost, and improved fall flows have provided conditions allowing adult migration throughout the fall season. Umatilla steelhead still exhibit a diverse age structure including multiple ages at smolt migration, multiple years of ocean residence and repeat spawning. The population rated at **moderate risk** because all pathways exist but there has been significant reduction in variability and changes in distribution.

B.1.b. Phenotypic variation.

We have no data to assess loss or substantial change is phenotypic traits, therefore we infer based on habitat changes. The changes in flow patterns and temperature profile within the Umatilla River and the mainstem Columbia River have likely resulted in reduced variation in adult and juvenile migration patterns. Juveniles have a much narrower window to successfully migrate out of the Umatilla in the spring because water temperatures increase earlier than historically. Even though flow enhancement has improved conditions for adult fall migration, the run-timing distribution is likely truncated from historic. Adults cannot enter the river in early fall in some years because of flow and temperature limitations. We have rated the Umatilla population at **moderate risk** because two or more phenotypic traits have changed.

B.1.c. Genetic variation

The genetics data for Umatilla steelhead indicate that there is significant within population variation between Umatilla steelhead and other populations in the MPG (Touchet, Walla Walla). In addition, the within population diversity shows no indication of impairment. The hatchery fish are similar to natural fish as expected, since they are offspring of natural fish. There are out-of-DPS spawners, primarily from Snake River stocks, spawning naturally in the Umatilla Basin. Given the degree of genetic variation the Umatilla population rated at **low risk** for this metric. Given that the genetics samples used in the analyses were collected from the mid-1980s, prior to significant hatchery influence, the genetic analyses needs to be updated with recent samples.

B.2.a. Spawner composition

(1) *Out-of-DPS spawners*. A significant number of out-of-DPS spawners enter the Umatilla River. Estimates of out-of-DPS spawners are based on expanded coded wire tagged recoveries

of hatchery fish at TMFD. From 1993-2004, out-of-DPS spawners have comprised from 1.8-9.7% (mean=4.8%) of the fish that arrived at TMFD. These strays are not selectively removed because they are not distinguishable from Umatilla Hatchery supplementation steelhead. Given the length of time of influence and the hatchery fraction, we have rated the Umatilla population at **moderate risk** for out-of-DPS spawners. This risk rating assumes strays were present at a similar rate for the past three generations.

(2) *Out-of-MPG spawners*. There have been few, if any, out-of-MPG within DPS spawners recovered in the Umatilla Basin, thus the rating is **very low** for this metric.

(3) *Out-of-population within MPG spawners*. There are two out of population within MPG hatchery programs which could provide stray fish to the Umatilla River, Lyons Ferry releases in the Walla Walla, and Touchet River hatchery fish. No strays from these two programs have been observed. The rating is **very low** for this metric.

(4) *Within-population hatchery spawners*. The Umatilla River population is supplemented annually with hatchery fish produced from wild broodstock collected at TMFD. The supplementation program has been ongoing since the late 1980's. Since 1993, Umatilla Hatchery fish have comprised an average of 29.4% of the natural spawning fish. We characterize this program as using best management practices based on the following:

- Most of the broodstock collected annually are wild fish.
- Mating protocols provide for a high number of family groups annually.
- There presently is no culling or grading of parr or smolts.
- Hatchery smolts are released in localized areas of the middle and upper mainstem.
- There does not appear to be any genetic differentiation between hatchery and natural fish.

Given that best practices are used, the average hatchery fraction is 29.4%, and the program has been underway for three generations, the rating is **moderate risk** for within population hatchery fish.

The overall risk rating for B.2.a. "spawner composition" is **high risk** because the out-of-DPS spawners and within-population hatchery proportions were both rated as moderate.

B.3.a. Distribution of population across habitat types

The intrinsic potential distribution encompasses seven ecoregions, four of which account for at least 10% of the distribution (Figure 6). There has been only one significant shift greater than 67% in the ecoregion distribution (Pleistocene Lake Basins). This population rates at **low risk**.

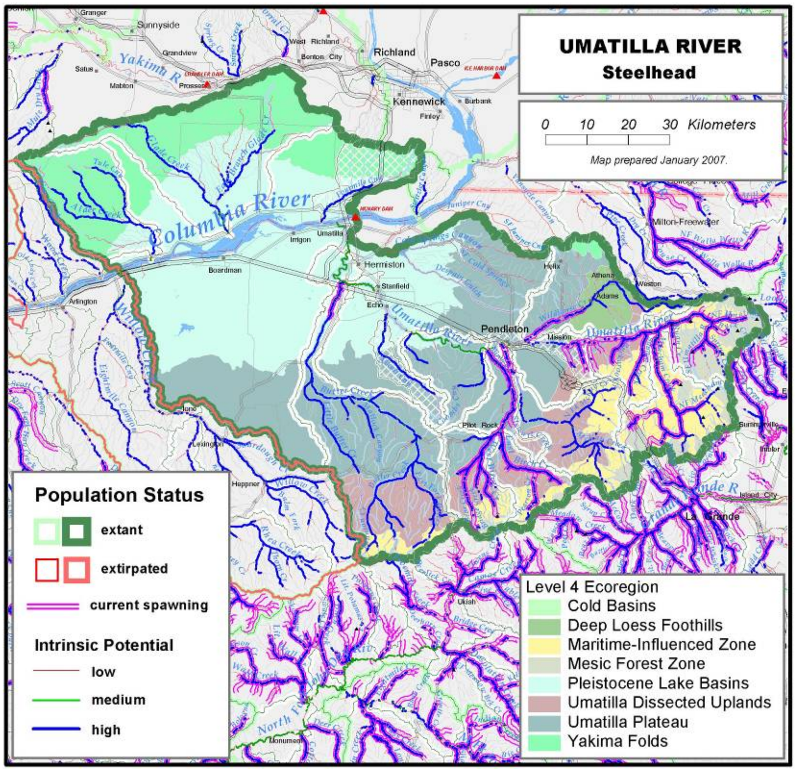

Figure 6. Umatilla River Summer Steelhead population spawning distribution across EPA level 4 ecoregions.

Table 3. Umatilla River Summer Steelhead population proportion of current spawning areas across EPA level 4 ecoregions.

Ecoregion	% of historical spawning area in this ecoregion (non-temperature limited)	% of currently occupied spawning area in this ecoregion (non-temperature limited)
Umatilla Plateau	32.4	27.0
Pleistocene Lake Basins	25.0	6.2
Yakima Folds	5.3	0.0
Deep Loess Foothills	2.7	1.2
Umatilla Dissected Uplands	15.3	19.3
Maritime-influenced Zone	17.7	42.9
Mesic Forest Zone	1.7	3.4

B.4.a. Selective change in natural processes or selective impacts

Hydropower system: The hydropower system and associated reservoirs impose some selective mortality on smolt outmigrants and upstream migrating adults. Selective mortality due to flow and temperature changes influences migration timing. The specific magnitude of selective mortality and the proportion of population that is affected are unknown. For the adult migration timing affects, the duration is multiple generations and the affect is intermittent as it does not occur each year. The proportion of the population affected is low resulting in low strength of selection. We consider adult migration timing to be highly heritable, thus the selective effects on adults are rated moderate risk with low strength of selection and high heritability. For selective mortality on smolt migration timing, the duration is multiple generations with the proportion of population affected low and the heritability low. We rated the smolt migration timing effect as low risk with low selection intensity low heritability. Overall the hydropower selectivity is rated at **moderate risk**.

Harvest: Recent harvest rates for Type-A steelhead in the Columbia River Mainstem are generally less than 10% annually. Although some harvest may be size selective for larger fish, the selective mortality would affect less than 2% of the total population. There is very limited tribal harvest of natural fish within the Umatilla Subbasin and impacts from the recreational fishery are incidental to hatchery fish harvest. There does not appear to be any selective mortality as a result of in-basin harvest. We rated this metric at **very low risk**.

Hatcheries: The Umatilla River summer steelhead hatchery program is operated to provide hatchery fish for harvest and to supplement natural production. Broodstock are collected at TMFD. Typically 100 naturally produced and 20 hatchery fish are collected for broodstock. Broodstock are collected representatively so that their run-timing, sex, and age of broodstock mimic that of the total run at TMFD. We are uncertain of the degree of substructure within the basin or if there are different characteristics between spawning aggregates in the basin. If life history characteristics differ between different aggregates, there is the possibility that collection of broodstock representing TMFD timing may be differentially impacting spawning aggregates. However, the broodstock removal does not appear to be selective at the population level thus we rated this metric at **very low risk**.

Habitat: There are two habitat changes, altered flow profiles and increased temperatures, which likely impose some selective mortality on pre-smolts, smolts, and adults. Mainstem summer temperatures are lethal in many reaches, and juveniles that leave tributary production areas and end up in the mainstem during summer likely suffer increased mortality. The proportion of population affected is low and the heritability is low, thus the juvenile selective impact is rated as low. Temperatures in the Umatilla River often reach stressful levels during the latter part of the smolt outmigration time period. The elevated temperatures likely impose higher mortality on the later migrating smolts. This affect has been ongoing for many generations. The proportion of the population affected is moderate and the heritability is low resulting in an overall rating of low for smolt impacts. Late summer and early fall flows are often low in the Umatilla River and adults entering the river early are likely subject to above normal mortality rates. For adults we rated the intensity of selection as low and the heritability as high resulting in an adult selectivity rating of moderate. The overall rating is **moderate risk** for habitat.

The combined selectivity rating for all four “H”s is **moderate risk**.

Spatial Structure and Diversity Summary

The combined integrated Spatial Structure/Diversity rating is moderate risk (Table 4) for the Umatilla River population. There has been significant reduction in spawner distribution relative to intrinsic potential distribution. This reduction has caused significant increases in gaps between spawning areas as well as disrupted continuity. Habitat changes have been significant in the Umatilla Basin resulting in changes to flow profiles and elevated temperatures. These changes have resulted in impacts to life history diversity and phenotypic trait variation. The out-of-DPS spawners in combination with local origin hatchery fish spawning naturally put the population at high risk for spawner composition. Hydrosystem effects and within basin habitat changes have likely resulted in selective mortality of specific components of juvenile and adult life stages resulting in a moderate risk rating.

Table 4. Umatilla River Summer Steelhead population spatial structure and diversity risk rating summary.

Metric	Risk Assessment Scores				
	Metric	Factor	Mechanism	Goal	Population
A.1.a	L (1)	L (1)	Mean=(0.33) Moderate Risk	Moderate Risk (0.33)	Moderate Risk
A.1.b	M (0)	M (0)			
A.1.c	M (0)	M (0)			
B.1.a	M (0)	M (0)	Moderate Risk (0)	Moderate Risk	
B.1.b	M (0)	M (0)			
B.1.c	L (1)	L (1)			
B.2.a(1)	M (0)	High Risk (-1)	High Risk (-1)		
B.2.a(2)	VL (2)				
B.2.a(3)	VL (2)				
B.2.a(4)	M (0)				
B.3.a	L (1)	L (1)	L (1)		
B.4.a	M (0)	M (0)	M (0)		

Overall Risk Rating

The Umatilla steelhead population does not currently meet the ICTRT recommended viability criteria because Abundance/Productivity and Spatial Structure/Diversity risks ratings are both moderate (Figure 7). However, the population does meet criteria for a “maintained” population. The 20-year delimited recruit per spawner point estimate is 1.50 with the lower end of the adjusted standard error above the 25% risk level, thus placing the productivity at low risk. The 10-year mean abundance of 1,472 is 98.1% of the minimum threshold of 1,500. Improvement in many of the Spatial Structure/Diversity metrics and a small increase in the average abundance will raise the population to viable status.

		Spatial Structure/Diversity Risk			
		Very Low	**Low**	**Moderate**	**High**
Abundance/ Productivity Risk	**Very Low** (<1%)	**HV**	**HV**	**V**	M*
	Low (1-5%)	**V**	**V**	**V**	M*
	Moderate (6 – 25%)	M*	M*	M* **Umatilla**	
	High (>25%)				

Figure 7. Umatilla River Summer Steelhead population risk ratings integrated across the four viable salmonid population (VSP) metrics. *Viability Key: HV – Highly Viable; V – Viable; M* – Candidate for Maintained; Shaded cells-- does not meet viability criteria (darkest cells are at highest risk).*

Umatilla River Summer Steelhead – Data Summary

Data type: Dataset reconstructed from dam counts

SAR: Averaged Deschutes, Umatilla, Snake River, and Upper Columbia Steelhead series

Table 5. Umatilla River Summer Steelhead population abundance and productivity data used for curve fits and R/S analysis. Bolded values were used in estimating the current productivity (Table 6).

Brood Year	Spawners	%Wild	Natural Run	Nat. Rtns	R/S	SAR Adj. Factor	Adj. Rtns	Adj. R/S
1981	**1,115**	**1.00**	**1,115**	**2,635**	**2.36**	**0.68**	**1799**	**1.61**
1982	**609**	**1.00**	**609**	**2,640**	**4.33**	**0.46**	**1207**	**1.98**
1983	**974**	**1.00**	**974**	**2,525**	**2.59**	**0.52**	**1322**	**1.36**
1984	1,998	1.00	1,998	1,943	0.97	0.65	1257	0.63
1985	2,732	1.00	2,732	1,559	0.57	0.46	716	0.26
1986	2,487	1.00	2,487	1,017	0.41	0.94	959	0.39
1987	2,911	1.00	2,911	1,144	0.39	2.18	2490	0.86
1988	2,201	0.93	2,050	1,573	0.71	0.99	1558	0.71
1989	2,179	0.84	1,841	1,105	0.51	0.96	1062	0.49
1990	1,301	0.96	1,247	873	0.67	2.83	2471	1.90
1991	**700**	**0.85**	**592**	**593**	**0.85**	**2.33**	**1384**	**1.98**
1992	2,118	0.90	1,915	1,380	0.65	1.88	2594	1.22
1993	1,572	0.74	1,165	713	0.45	1.18	842	0.54
1994	**1,074**	**0.79**	**847**	**885**	**0.82**	**1.07**	**948**	**0.88**
1995	1,298	0.60	783	1,154	0.89	1.23	1414	1.09
1996	1,811	0.66	1,194	2,975	1.64	1.03	3070	1.70
1997	2,215	0.41	914	2,210	1.00	0.76	1687	0.76
1998	1,529	0.50	771	3,836	2.51	0.49	1880	1.23
1999	1,595	0.64	1,020	1,071	0.67	0.52	554	0.35
2000	2,621	0.77	2,030	2,584	0.99	1.00	2584	0.99
2001	3,353	0.73	2,444					
2002	5,172	0.68	3,542					
2003	2,822	0.71	2,015					
2004	3,109	0.64	2,003					

Table 6. Umatilla River Summer Steelhead population geometric mean abundance and productivity estimates (values used for current productivity and abundance are shown in boxes).

	R/S measures				Lambda measures		Abundance
	Not adjusted		SAR adjusted		Not adjusted		Nat. origin
delimited	median	75% threshold	median	75% threshold	1989-2000	1981-2000	geomean
Point Est.	1.24	1.79	1.14	1.50	1.07	1.06	1472
Std. Err.	0.24	0.33	0.19	0.15	0.02	0.06	0.22
count	10	5	10	5	12	20	10

Table 7. Umatilla River Summer Steelhead population stock-recruitment cure fit parameter estimates. Biologically unrealistic or highly uncertain values are highlighted in grey.

	Not adjusted for SAR							Adjusted for SAR						
SR Model	a	SE	b	SE	adj. var	auto	AICc	a	SE	b	SE	adj. var	auto	AICc
Rand-Walk	0.94	0.14	n/a	n/a	0.27	0.60	44.5	0.89	0.12	n/a	n/a	0.31	0.31	40.3
Const. Rec	1512	174	n/a	n/a	n/a	n/a	34.8	1438	147	n/a	n/a	n/a	n/a	30.2
Bev-Holt	22.07	116.06	1587	446	0.21	0.44	37.5	8.48	15.93	1625	425	0.20	-0.15	32.7
Hock-Stk	1.92	0.70	806	310	0.21	0.45	38.1	1.98	0.64	735	249	0.20	-0.18	32.8
Ricker	2.70	0.88	0.00060	0.00017	0.22	0.45	38.0	2.35	0.69	0.00055	0.00016	0.21	-0.14	33.4

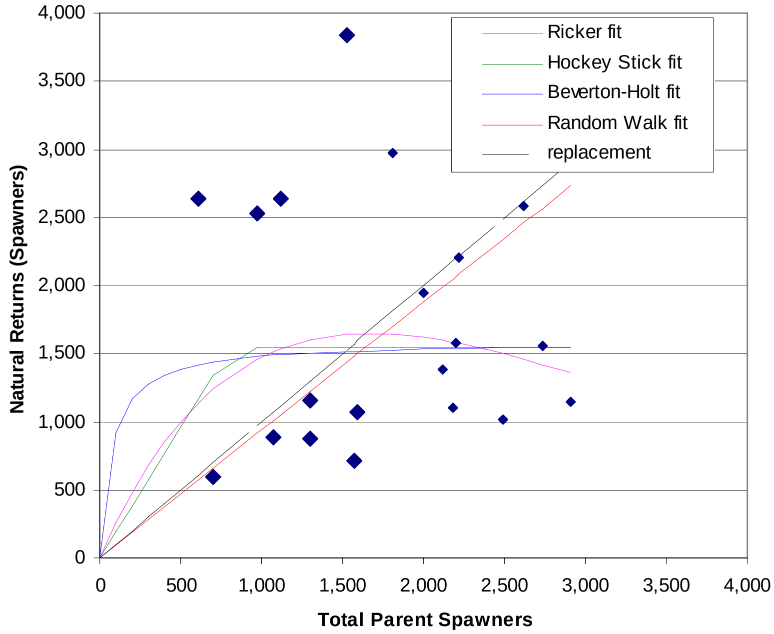

Figure 8. Umatilla River Summer Steelhead population stock recruitment curves. Bold points were used in estimating the current productivity. Data were not adjusted for marine survival.

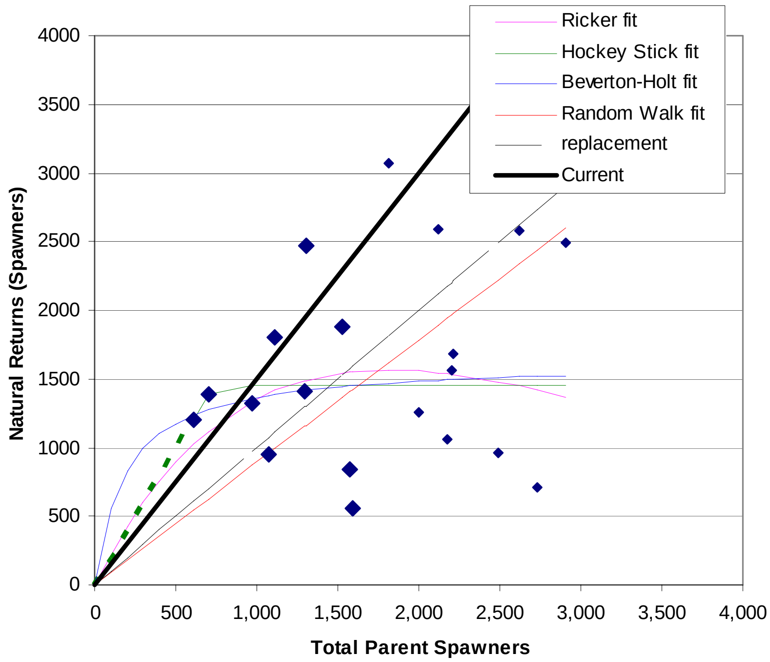

Figure 9. Umatilla River Summer Steelhead population stock recruitment curves. Bold points were used in estimating the current productivity. Data were adjusted for marine survival.